
Protecting 30% of the planet for nature: costs, benefits and economic implications

**Working paper analysing the economic implications of the proposed 30% target for
areal protection in the draft post-2020 Global Biodiversity Framework**

Anthony Waldron¹; Vanessa Adams²; James Allan³; Andy Arnell⁴; Greg Asner⁵; Scott Atkinson⁶; Alessandro Baccini⁷; Jonathan EM Baillie⁸; Andrew Balmford¹; J Austin Beau⁹; Luke Brander¹⁰; Eduardo Brondizio¹¹; Aaron Bruner¹²; Neil Burgess⁴; K Burkart¹³; Stuart Butchart¹⁴; Rio Button¹⁵; Roman Carrasco¹⁶; William Cheung¹⁷; Villy Christensen¹⁸; Andy Clements¹⁹; Marta Coll²⁰; Moreno di Marco²¹; Marine Deguignet⁴; Eric Dinerstein²²; Erle Ellis²³; Florian Eppink²⁴; Jamison Ervin²⁵; Anita Escobedo²⁶; John Fa²⁷; Alvaro Fernandes-Llamazares²⁸; Sanjiv Fernando²²; Shinichiro Fujimori²⁹; Beth Fulton³⁰; Stephen Garnett³¹; James Gerber³²; David Gill³³; Trisha Gopalakrishna³⁴; Nathan Hahn²²; Ben Halpern³⁵; Tomoko Hasegawa³⁶; Petr Havlik³⁷; Vuokko Heikinheimo²⁸; Ryan Heneghan³⁸; Ella Henry³⁹; Florian Humpenoder⁴⁰; Harry Jonas⁴¹; Kendall Jones⁴²; Lucas Joppa⁴³; A.R. Joshi⁴⁴; Martin Jung³⁷; Naomi Kingston⁴; Carissa Klein⁴⁵; Tamas Krisztin³⁷; Vicky Lam⁴⁶; David Leclere³⁹; Peter Lindsey⁴⁷; Harvey Locke⁴⁸; TE Lovejoy⁴⁹; Philip Madgwick⁵⁰; Yadvinder Malhi³⁴; Pernilla Malmer⁵¹; Martine Maron⁵²; Juan Mayorga⁵³; Hans van Meijl⁵⁴; Dan Miller⁵⁵; Zsolt Molnar⁵⁶; Nathaniel Mueller⁵⁷; Nibedita Mukherjee¹; Robin Naidoo⁵⁸; Katia Nakamura⁵⁹; Prakash Nepal⁶⁰; RF Noss⁶¹; Beth O'Leary⁶²; D Olson⁶³; Juliano Palcios Abrantes⁶⁴; Midori Paxton⁶⁵; Alexander Popp⁴⁰; Hugh Possingham⁶⁶; Jeff Prestemon⁶⁷; April Reside²¹; Catherine Robinson³⁰; John Robinson⁶⁸; Enric Sala⁶⁹; Kim Scherrer⁷⁰; Mark Spalding⁶⁶; Anna Spenceley⁷¹; Jeroen Steenbeck⁷²; Elke Stehfest⁷³; Bernardo Strassburg⁷⁴; Rashid Sumaila¹⁷; Kirsty Swinnerton⁷⁵; Jocelyne Sze⁷⁶; Derek Tittensor⁴; Tuuli Toivonen⁷⁷; Alejandra Toledo⁷⁸; Pablo Negret Torres²¹; Willem-Jan Van Zeist⁷³; James Vause⁴; Oscar Venter⁷⁹; Thais Vilela⁸⁰; Piero Visconti¹; Carly Vynne²²; Reg Watson⁸¹; James Watson²¹; Eric Wikramanayake⁸²; Brooke Williams²¹; Brendan Wintle⁸³; Stephen Woodley⁸⁴; Wenchao Wu⁸⁵; Kerstin Zander⁸⁶; Yuchen Zhang¹⁶; YP Zhang⁸⁷

Author Affiliations/Institutions: ¹Conservation Science Group, Department of Zoology, University of Cambridge, The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ²Discipline of Geography and Spatial Sciences, University of Tasmania, Hobart, Tasmania, Australia; ³University of Amsterdam, Institute for Biodiversity and Ecosystem Dynamics, Amsterdam, NL; ⁴UN Environment World Conservation Monitoring Centre, 219 Huntingdon Road, Cambridge, CB3 0DL, United Kingdom; ⁵Center for Global Discovery and Conservation Science, TEMPE Campus, 1001 S. McAllister Ave, Tempe, AZ 85281, USA; ⁶The Surf Conservation Partnership, Conservation International, 2011 Crystal Drive, Suite 600, Arlington, VA 22202, USA; ⁷Woods Hole Research Center, 149 Woods Hole Road, Falmouth, MA, 02540-1644, USA; ⁸Department of Zoology, University of Oxford, Zoology Research and Administration Building, 11a Mansfield Road, Oxford OX1 3SZ, UK; ⁹Research Institute for the Environment and Livelihoods, Charles Darwin University; Water Flagship, Commonwealth Science and Industry Research Organisation, Australia; ¹⁰Institute for Environmental Studies, VU University Amsterdam, Amsterdam, Netherlands; ¹¹Department of Anthropology, Indiana University, Bloomington, IN, USA; ¹²Conservation Agreement Fund, 14212 Northwyn Drive, Silver Spring, MD 20904, USA; ¹³One Earth; ¹⁴Birdlife International, The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ¹⁵University of Cape Town, Cape Town, South Africa; ¹⁶Department of Biological Sciences, National University of Singapore, 14 Science Drive 4, Singapore 117543; ¹⁷Institute for the Oceans and Fisheries, Faculty of Science, Vancouver Campus, The University of British Columbia, AERL, 2202 Main Mall, Vancouver, BC Canada V6T 1Z4; ¹⁸Institute for the Oceans and Fisheries, Biodiversity Research Centre, 2212 Main Mall, Vancouver, BC Canada V6T 1Z4; ¹⁹BTO, The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ²⁰Institute of Marine Science (ICM-CSIC), Barcelona, Spain; ²¹Centre for Biodiversity and Conservation Science, The University of Queensland, Brisbane, QLD, 4072, Australia; ²²Resolve, 1255 23rd Street NW, Suite 275, Washington, DC 20037, USA; ²³University of Maryland, Baltimore County, 1000 Hilltop Circle, Baltimore, MD 21250, USA; ²⁴Institute for Biodiversity - Network e.V. (ibn), Nußbergerstr. 6a, Regensburg, Germany; ²⁵Global Programme on Nature for Development, UNDP, New York, USA; ²⁶Conservation Strategy Fund, Lima, Peru; ²⁷Manchester Metropolitan University, All Saints Building, Manchester, M15 6BH, United Kingdom; ²⁸Helsinki Institute of Sustainability Science (HELSUS), Global Change and Conservation Lab, PL 65 (Viikinkaari 1), 00014, Finland; ²⁹National Institute for Environmental Studies (NIES), Center for Social and Environmental Systems Study, 16-2, Onogawa, Tsukuba, JAPAN; ³⁰CSIRO, GPO BOX 1538, Hobart, TAS 7001 AUSTRALIA; ³¹Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory, Australia; ³²Institute on the Environment, University of Minnesota, 1954 Buford Avenue, Saint Paul, MN 55108, USA; ³³Duke Univ Marine Lab, 135 Duke Marine Lab Road, Beaufort, NC 28516, USA; ³⁴School of Geography and the Environment, University of Oxford, South Parks Road, Oxford OX1 3QY, UK; ³⁵Bren School of Environmental Science & Management, 2400 Bren Hall, University of California, Santa Barbara CA 93106-5131; ³⁶Ritsumeikan BKC Campus, 1-1-1 Noji-higashi, Kusatsu, Shiga 525-8577 Japan; ³⁷International Institute for Applied Systems Analysis (IIASA) - Schlossplatz 1 - A-2361 Laxenburg, Austria; ³⁸Institut de Ciència i Tecnologia Ambientals (ICTA-UAB), Universitat Autònoma de Barcelona, 08193 Barcelona, Spain; ³⁹Cambridge University Zoology Dept, Cambridge, UK CB2; ⁴⁰Potsdam Institute for Climate Impact Research (PIK), Telegraphenberg A56 / 104, 14412 Potsdam, Germany; ⁴¹Natural Justice, 63 Hout Street, Mercantile Building, Cape Town, 8000, South Africa; ⁴²Wildlife Conservation Society, The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ⁴³The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ⁴⁴University of Minnesota, 135 B Skok Hall, 2003 Upper Buford Circle, St. Paul, MN 55108, USA; ⁴⁵School of Earth and Environmental Sciences, The University of Queensland, Brisbane, QLD, 4072, Australia; ⁴⁶Nippon Foundation-Nereus Program and Changing Ocean Research Unit, Institute for the Oceans and Fisheries, The University of British Columbia, Vancouver, BC, Canada; ⁴⁷Wildlife Conservation Network, 4 Morningside Drive, Mount Pleasant, Harare, Zimbabwe; ⁴⁸World Commission on Protected Areas, International Union for Conservation of Nature, 1196 Gland, Switzerland; ⁴⁹Biology Department, George Mason University, 4400 University Drive MS3E, Fairfax, VA, 22030, USA; ⁵⁰Centre, Bracknell, Berkshire, RG42 6EY, UK; ⁵¹SwedBio, Stockholm Resilience Centre, Stockholm University, Kråftriket 2B, SE-106 91 Stockholm; ⁵²School of Earth and Environmental Sciences and Centre for Biodiversity and Conservation Science, University of Queensland, St Lucia, Queensland, Australia; ⁵³University of California, Santa Barbara, Santa Barbara, CA, USA; ⁵⁴Wageningen economic Research, Wageningen University and Research Centre, The Hague, Netherlands; ⁵⁵S-406 Turner Hall, 1102 S. Goodwin, Urbana Illinois 61801, USA; ⁵⁶Centre for Ecological Research Institute of Ecology and Botany, Magyar Tudományos Akadémia, H-2163 Vácraátót, Hungary; ⁵⁷Department of Earth System Science, University of California, Irvine, Irvine, California, USA; ⁵⁸World Wildlife Fund, Washington, DC, USA; ⁵⁹Department of Natural Resources and Environmental Sciences, University of Illinois, Urbana, IL 61801, United States; ⁶⁰USDA Forest Service, Forest Products Laboratory, Madison, WI, USA; ⁶¹Florida Institute for Conservation Science, Sarasota, FL, USA; ⁶²Department of Environment and Geography, Wentworth Way, University of York, Heslington, York, YO10 5NG, UK; ⁶³WWF Hong Kong, 15/F, Manhattan Centre, 8 Kwai Cheong Road, Kwai Chung, New Territories, HK; ⁶⁴The University of British Columbia, AERL, 2202 Main Mall, Vancouver, BC Canada V6T 1Z4; ⁶⁵UNDP Global Environmental Finance Unit, 220 East 42nd St. Room 2036, New York City, New York, USA; ⁶⁶The Nature Conservancy, 4245 North Fairfax Drive, Suite 100, Arlington, Virginia 22203-1606; ⁶⁷USDA, Southern Research Station, 200 W.T. Weaver Blvd., Asheville, NC 28804-3454, USA; ⁶⁸Wildlife Conservation Society, New York, NY, USA; ⁶⁹National Geographic Society, 1145 17th Street NW, Washington, DC 20036, USA; ⁷⁰Department of Earth Sciences, Uppsala University, Uppsala, Sweden; ⁷¹IUCN WCPA Tourism and Protected Areas Specialist Group (TAPAS Group); ⁷²Institute of Marine Science, Ecopath International Initiative, 08003 Barcelona, Spain; ⁷³PBL Netherlands Environmental Assessment Agency, Postbus 30314, The Hague, 2500 GH, The Netherlands; ⁷⁴Rio Conservation and Sustainability Science Centre, Department of Geography and the Environment, Pontifícia Universidade Católica, Rio de Janeiro, Brazil; ⁷⁵Kent Wildlife Trust, Tyland Barn, Chatham Rd, Sandling, Maidstone ME14 3BD, UK; ⁷⁶The University of Sheffield, Western Bank, Sheffield, S10 2TN, UK; ⁷⁷University of Helsinki, PL 4 (Yliopistonkatu 3), 00014, Finland; ⁷⁸Independent; ⁷⁹University of Northern British Columbia, 3333 University Way, Prince George, BC Canada V2N 4Z9; ⁸⁰Conservation Strategy Fund, 1875 Connecticut Ave, NW, 10th floor, Washington, DC 20009, USA; ⁸¹Institute for Marine and Antarctic Studies, University Tasmania, Taroona, Tas., 7001 Australia; ⁸²Director, Wildlife and Wetlands, WWF Hong Kong; ⁸³School of Bioscience, University of Melbourne, Building 122, Melbourne, Victoria, 3010 Australia; ⁸⁴IUCN, The David Attenborough Building, Pembroke Street, Cambridge, CB2 3QZ, United Kingdom; ⁸⁵Center for Social and Environmental Systems Research, National Institute for Environmental Studies, Tsukuba, Japan; ⁸⁶Northern Institute, Charles Darwin University, Ellengowan Drive, Brinkin, 0909, Darwin, NT, Australia; ⁸⁷Kunming Institute of Zoology, the Chinese Academy of Sciences No. 32 Jiaochang Donglu, Kunming, Yunnan, 650223, P.R. China

TABLE OF CONTENTS

EXECUTIVE SUMMARY (Bulleled Abstract)	5
<u>Figure 1. All-sector economic output projected when implementing (or not implementing) the 30% target</u>	6
<u>Figure 2. Effects of implementing the 30% target on individual sector outputs</u>	7
<u>Figure 3. Revenue growth projected when implementing (or not implementing) the 30% target</u>	7
<u>Figure 4. Loss and recovery of fisheries catch and fisheries revenue</u>	8
<u>Figure 5. Costs and benefits for the PA sector only</u>	8
EXECUTIVE SUMMARY	9
Findings of the report	11
FULL REPORT	14
Introduction: The 30% Target for Protected Areas and OECMs	14
Purpose and Framework of this Report: Benefits and Costs	16
Creating Different Scenarios for how the 30% Target might be Implemented	17
<u>Table 1. Terrestrial and marine protected-area scenarios used to explore the economic implications of the 30% target</u>	18
<u>Table 2. Scenarios for the full 30% target, including both terrestrial and marine protected areas</u>	19
Estimating Total and Sector-specific Economic Outputs for the Different Policy Options	19
Methods (Financial Analysis)	20
PA/Nature Sector Outputs/Revenues	20
Agricultural Sector Outputs/Revenues	21
Forestry Sector Outputs/Revenues	21
Fisheries Sector Outputs/Revenues and Catch	22

Results (Financial Analysis)	23
Financial Benefits of Implementing the 30% Target	23
Breakdown of Outputs/Revenues and Opportunity Costs By Sector	23
<i>PA/Nature</i>	23
<i>Agriculture</i>	23
<i>Forestry</i>	24
<i>Fisheries</i>	24
Revenue Growth in the Nature Sector, Compared to Other Sectors and Overall GDP	25
Broad-sense Economic and Non-monetary Values	26
Costs and Investment Needs for the 30% Target	27
Methods: Budget Needs (costs)	27
Results: Budget Needs (costs) and Comparison with Current Spending	29
Synthesis: Putting Together the Benefits and Costs of Investing in the 30% Target	30
Distribution of costs and benefits, development assistance, and cross-subsidy	32
Table 3. Benefits and costs of scenarios to expand protected areas (2050)	33
 AUTHOR CONTRIBUTIONS	 35
 FUNDING ACKNOWLEDGMENTS	 36
 METHODS APPENDIX:	
Protecting 30% of the Planet for Nature: Costs, Benefits and Economic Implications	37
Revenues/Sector Outputs	37
<i>PA/Nature Sector</i>	37
<i>Agricultural Sector</i>	41
<i>Forestry Sector</i>	42
<i>Fisheries Sector</i>	42
<i>Avoided-cost and Ecosystem Service Values</i>	43
<i>A Note on Discounting Versus Forward Planning</i>	45
Costs/Investment Needs	45
<i>Terrestrial Protected Areas: Annual Budget Needs</i>	46
<i>Marine Protected Areas: Annual Budget Needs</i>	48
<i>Establishment Costs</i>	49
<i>Current Global Spending on Protected Areas</i>	49
 REFERENCES	 50

EXECUTIVE SUMMARY (Bulleted Abstract)

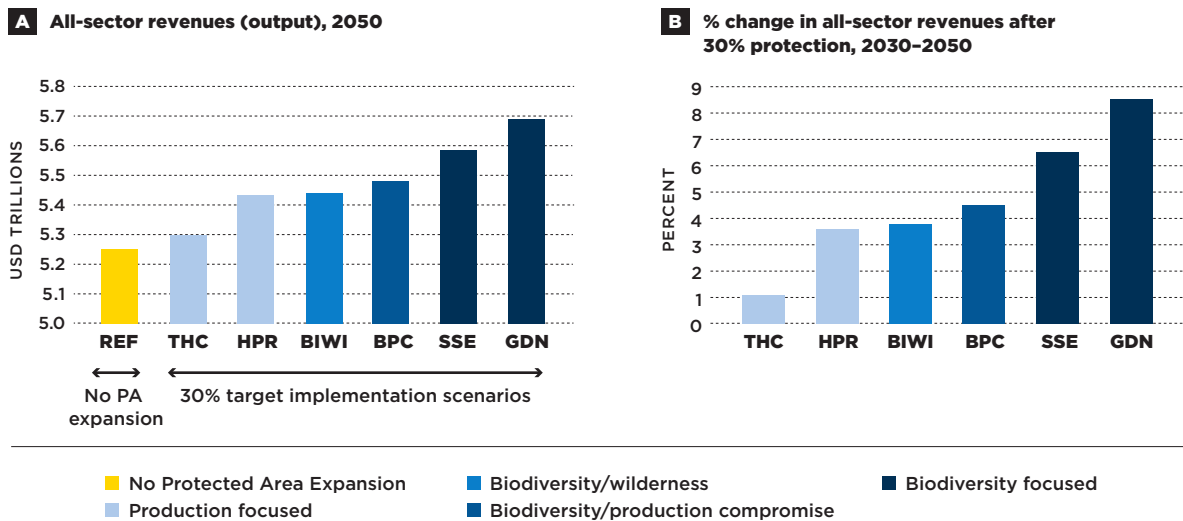
- The World Economic Forum now ranks biodiversity loss as a top-five risk to the global economy, and the draft post-2020 Global Biodiversity Framework proposes an expansion of conservation areas to 30% of the earth's surface by 2030 (hereafter the "30% target"), using protected areas (PAs) and other effective area-based conservation measures (OECMs).
- Two immediate concerns are how much a 30% target might cost and whether it will cause economic losses to the agriculture, forestry and fisheries sectors.
- Conservation areas also generate economic benefits (e.g. revenue from nature tourism and ecosystem services), making PAs/Nature an economic sector in their own right.
- If some economic sectors benefit but others experience a loss, high-level policy makers need to know the net impact on the wider economy, as well as on individual sectors.
- The current report, based on the work of over 100 economists/scientists, analyses the global economic implications of a 30% PA target for agriculture, forestry, fisheries, and the PA/nature sector itself. (OECMs were only defined by the CBD in 2018, too recently to economically model, but we include a qualitative treatment of them.)
- We carried out two analyses: a global *financial* one (concrete revenues and costs only); and a tropics-focused *economic* one (including non-monetary ecosystem service values), for multiple scenarios of how a 30% PA target might be implemented.
- Our *financial* analysis showed that expanding PAs to 30% would generate higher overall output (revenues) than non-expansion (an extra \$64 billion-\$454 billion per year by 2050). (Figure 1-2).
- In the *economic* analysis, only a partial assessment was possible, focusing on forests and mangroves. For those biomes alone, the 30% target had an avoided-loss value of \$170-\$534 billion per year by 2050, largely reflecting the benefit of avoiding the flooding, climate change, soil loss and coastal storm-surge damage that occur when natural vegetation is removed. The value for all biomes would be higher.
- Implementing the proposal would therefore make little initial difference to total (multi-sector) economic output, although a modest rise in gross output value is projected.
- The main immediate difference between expansion and non-expansion is therefore in broader economic/social values. Expansion outperforms non-expansion in mitigating the very large economic risks of climate change and biodiversity loss (Figure 5). The 30% target would also increase by 63%-98% the area recognised as Indigenous Peoples' and local communities' land-based nature stewardship contribution (within appropriate rights and governance frameworks).
- Economic growth in the PA/nature sector (at 4-6%) was also many times faster than the 1% growth expected in competing sectors (Figure 3). Marine expansion restores growth to fisheries (after a shock) but non-expansion leads to a mid-term contraction (Figure 4).

- The annual investment needed for an expanded (30%) PA system is \$103 – \$178 billion¹. This figure includes \$68 billion for the existing system, of which only \$24.3 is currently spent. (Underfunded systems lose revenue, assets, carbon and biodiversity).
- Most of the investment need is in low- and middle-income countries (LMICs). These often have a competitive asset advantage in terms of natural areas, but they may need international support to capitalise on that opportunity. Otherwise, growing the PA sector could also entrench global economic inequalities.
- Benefits and costs also accrue to different stakeholders at smaller (e.g. local) scales, making welfare distribution a challenge that needs addressing.

FIGURE 1.

All-sector economic output projected when implementing (or not implementing) the 30% target

A. Projected “all-sector” economic output in 2050. (All-sector output is the sum of all outputs across the four economic sectors affected by the 30% target: PA/Nature, Agriculture, Forestry, and Fisheries). All six scenarios for PA expansion are shown and compared to no-PA-expansion: REF = no-expansion scenario, THC = Three Conditions, HPR = Harsh Political Reality, BIWI = Biodiversity/Wilderness Consensus, BPC = Biodiversity/Production Compromise, SSE = Save Species from Extinction, GDN = Global Deal for Nature. **B.** Projected % increase/decrease in the all-sector output if 30%-target scenarios are implemented (by 2050).



¹ Excluding the budget estimate for the Global Deal for Nature scenario, see Full Report for details.

FIGURE 2.

Effects of implementing the 30% target on individual sector outputs

A. Change in individual sector output values (USD billions in 2050) projected to occur if the 30% target is implemented.
B. Projected % changes in agriculture-only net output value in 2050, compared to the no-PA-expansion scenario.
 See Figure 1 caption for scenario names.

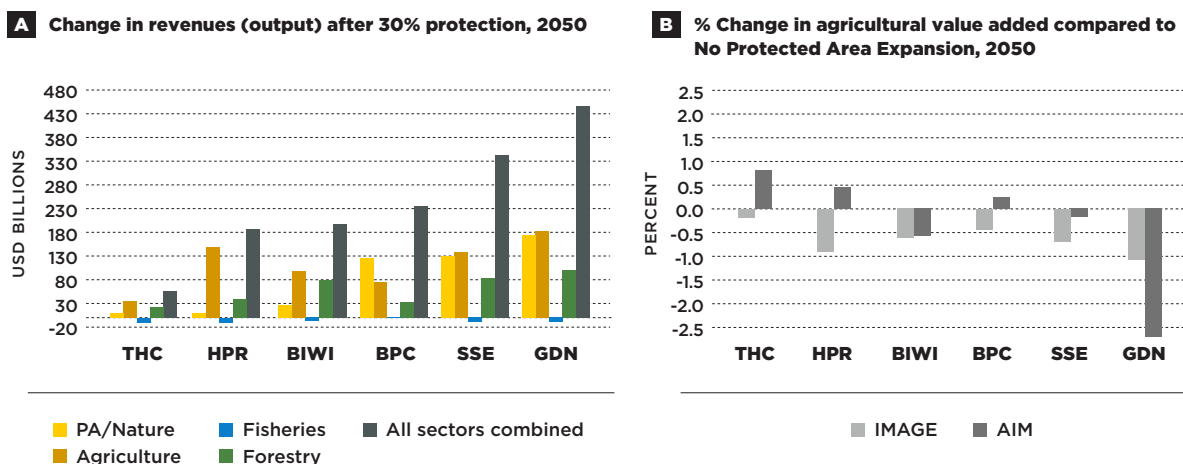


FIGURE 3.

Revenue growth projected when implementing (or not implementing) the 30% target

Projected annual growth rates in outputs per sector (2030-2050), showing the main scenario categories (production focus = HPR, biodiversity focus = SSE). CAGR = compound annual growth rate.

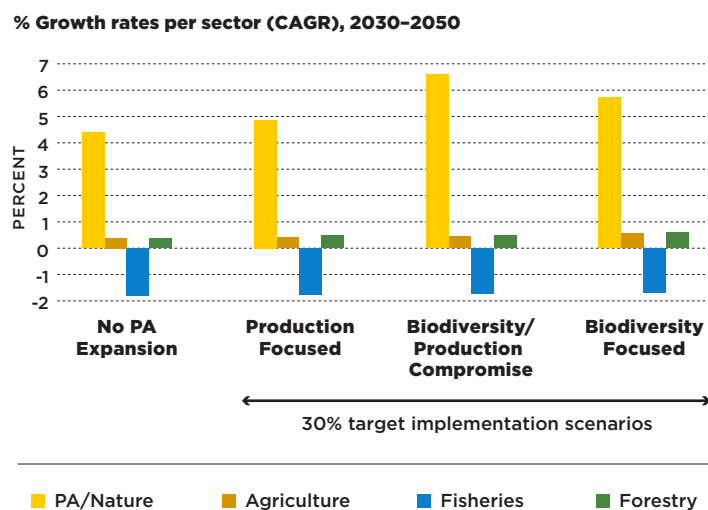


FIGURE 4.

Loss and recovery of fisheries catch and fisheries revenue

A. Change in global fisheries revenue over time (BOATS model). Note how 2030 revenue is always lower than in 2020, with or without MPA expansion, but expansion creates a bigger drop; how revenue increases in 2030-2040, irrespective of scenario; but how in 2040-2050, revenue only continues to increase if MPA expansion was implemented in 2020-2030. **B.** Expansion-driven loss and recovery in *relative* fisheries catch biomass. Blue bars show initial (2020-2030) loss in *absolute* catch biomass, where <100% shows catch reduction relative to the no-expansion baseline (EcoOcean model). Orange and grey bars show different rates of biomass *recovery* after 2030, where >100% indicates better *recovery* than the no-expansion baseline.

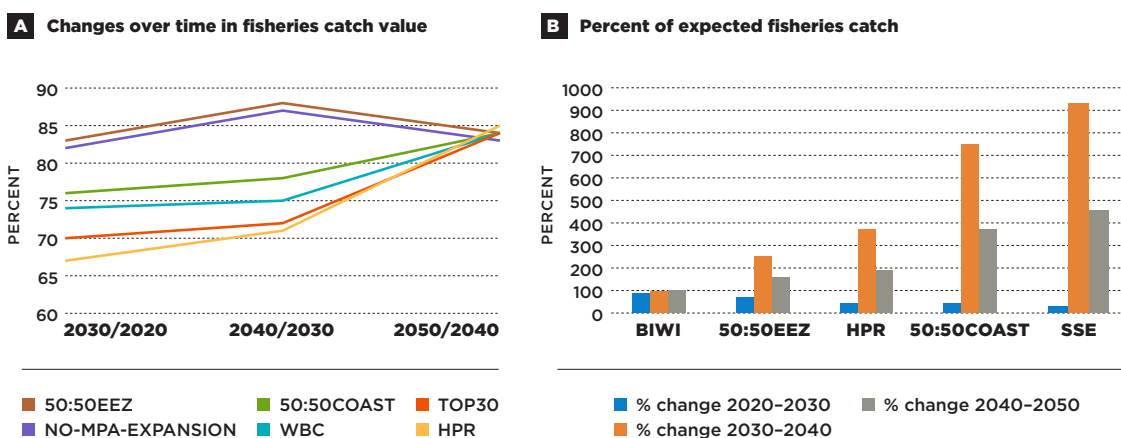
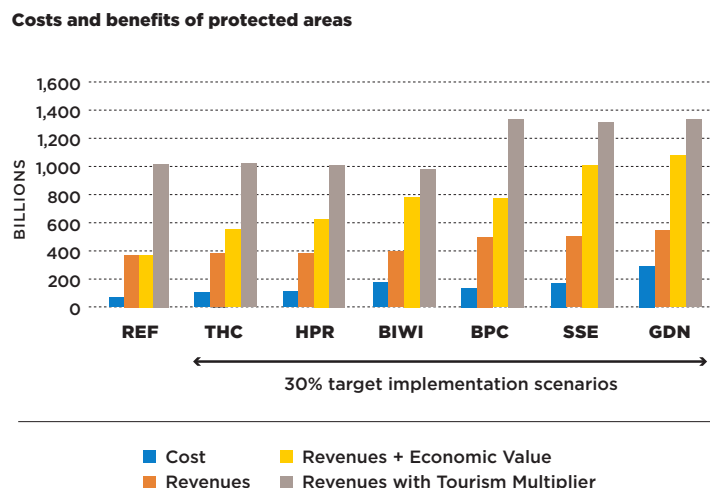


FIGURE 5.

Costs and benefits for the PA sector only

Comparing the public investment costs and the revenues/economic benefits in the PA/nature sector alone (ignoring other sectors). We show annual costs and three types of benefit in 2050: (i) the direct *financial* revenues; (ii) sum of *financial* revenues plus annual *economic* values; (iii) *financial* revenues that use a naïve universal multiplier (3.2) to capture the additional value of PA visits to supply chains and the downstream economy. See Figure 1 for scenario-name abbreviations.



EXECUTIVE SUMMARY

The World Economic Forum (WEF) and Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) have both identified biodiversity loss as one of the main threats to global economic prosperity^{1,2}. In particular, this high-level warning specifies that any further loss of natural habitats and biodiversity will cause extensive and costly flooding, climate change, disease emergence and ill health, clean water shortages, loss of crop pollination, decline in productivity, and numerous other risks^{1,2}. All of these negative outcomes are the consequence of degrading the natural infrastructure that supports human economic activity and wellbeing. One of the main policy instruments to slow the loss of biodiversity and the degradation of nature is the creation of protected or conservation areas (simple examples being a Nature Reserve or Marine Reserve)³⁻⁶. Currently, ~16% of the land and 7.4% of the ocean is in areas designated or proposed for protection (although only 2.5% of the ocean is in highly/fully protected areas)^{7,8}. This level of protection is widely acknowledged as being inadequate to achieve biodiversity protection goals^{7,9-12}. One of the headline proposals for the 15th meeting of the Conference of Parties to the CBD, and Action Target 2 of the draft post-2020 Global Biodiversity Framework¹³, is to increase the area covered by protected areas (PAs) and other effective area-based conservation measures (OECMs) to 30% of the planet by 2030, including both land and water protection.

This proposal to greatly expanded protected areas is likely to cause concern about potential losses of production and revenue, especially in agriculture, forestry and fisheries. It is also likely to raise concerns about the implied need for (potentially large) increases in public spending on the environment. However, protected areas also have multiple economic benefits, including nature tourism income, the provision of clinics, education and other forms of support to local communities, improved health outcomes, and avoidance of catastrophic losses due to the degradation of nature (the essence of the World Economic Forum's risk warning)^{2,14-17}. The benefits of expanding PAs therefore need to be weighed against the costs. The present report was commissioned to calculate the economic costs and benefits of implementing the 30% target. Over 100 scientists and economists contributed to the report, generating approximately half a billion pieces of data, and bringing together the fields of ecology, agriculture, fisheries, finance, tourism, anthropology, indigenous studies, ethnobiology, and climate science. Despite this background, the report itself aims to briefly summarise the main findings.

We adopt a total-economy or "multi-sector" framework, in which multiple economic and social interest-groups compete for the use of land or ocean. If a decision allows one sector to expand onto its first choice of land or sea area, other sectors have to expand or redistribute activity onto their second-best option, generating a tapestry of multiple benefits and costs. At the highest level, decision makers would typically take an overview of all those sectoral benefits and costs, in order to make a more holistic judgment of what is best for the economy, or for social and political priorities. The pattern of costs and benefits also evolves over time, and so a predictive framework is needed, projecting the outcomes in the near future. Indeed, it would take time for the proposed 2030 target to be debated, adopted and implemented, and at least a decade more for economies to adjust to such a major change in land-use and ocean-use policy. To examine the outcomes that would follow from implementing a target with a date of 2030, we therefore focus on the outcomes for 2050 (the target year for the CBD mission and the Sustainable Development Goals), with an interim value for 2040.

Since terrestrial PA expansion would generally occur on natural land (and MPA expansion in biologically important ocean areas), the economic sectors most widely impacted would be agriculture, forestry and fisheries. However, we reiterate that protected areas themselves form an important part of the modern global economy, and so should be regarded as an economic sector in their own right. Our multi-sector analysis therefore covers four major economic sectors in total: (1) agriculture, (2) forestry, (3) fisheries, and (4) the PA/nature sector (which includes a substantial portion of the global tourism sector, alongside several other income flows). Some of the 30% target could be achieved by implementing Other Effective Area-Based Conservation Measures (OECMs)^{18,19}. However, OECMs were only defined by the CBD in 2018 and as yet, there is insufficient information on how they might differ from PAs in terms of benefits and costs, so we focus our analysis on protected areas (PAs) in the more traditional sense e.g. National Parks and marine parks. We nevertheless discuss OECMs qualitatively in the Full Report.

We compare expected outcomes for a number of policy options. The first option is the no-PA-expansion scenario, in which the proposal is not implemented. We then explore number of alternative scenarios, all representing different possible implementations of the 30% target. A total of six terrestrial and five marine alternative scenarios were studied, but they can be grouped into three main categories, relating to the underlying philosophy. The first category (referred to as “Production-focused”) gives priority to the traditional productive sectors such as agriculture and fisheries, and then positions new protected areas where they do not conflict with those sectors. The second category, “Biodiversity-focused”, instead gives priority to biological conservation, placing protected areas where they would be most valuable for a range of biodiversity components. The third category, “Biodiversity/Production Compromise”, contains scenarios that represent a halfway compromise between biodiversity needs and the traditional productive sectors.

We begin by analysing the **financial** outcomes for each policy option. Since the first concern with PAs is that they will lower the level of economic output, across multiple sectors, we focus principally on output and in particular, the total output (sometimes referred to as the total revenue generated) across all the sectors addressed. Where feasible, we also explore net output values (revenues minus costs). Total economic output is made up of both the direct revenues generated by the products and services (e.g. sale of agricultural commodities or expenditure by nature tourists), and a ‘multiplier’ effect²⁰⁻²² that reflects how supply chain industries (e.g. for tourism needs or agricultural production) will also grow as a result, and how the additional revenues in the economy then lead to greater ‘downstream’ economic activity as they are re-spent on other goods and services. Our analysis mostly focused on the direct output, which is smaller, but the multiplier effect is important and we also briefly explore its implications.

However, many economically important functions are not designed to generate revenue. First, there is the value of ‘avoided cost’ – where the aim is not to increase spending (revenues) in the consumer economy, but rather to avoid (defend against) the risk of having to spend potentially large sums on disaster recovery. For example, spending on flood defence, hurricane mitigation, public health or the military all defend against potentially catastrophic and costly risks, as well as preserving a way of life and a set of values. Similarly, the World Economic Forum’s warning about biodiversity loss refers to the need to avoid potentially very large costs of failing to protect nature from further degradation. Classic examples of nature-loss risks include large-scale flooding, ill health and epidemics, lower food security, climate change, destruction of coastlines, and wildfires².

Beyond avoided costs, some of the values of nature are fully non-monetary, either because the value of protecting them is not yet financially recognised (e.g. many administrative areas have not yet given economic recognition to the water purification services provided by protected forests²³), or because a market value would be completely inappropriate (e.g. the cultural and spiritual value of preserving a tiger or a sacred forest). To avoid confusion in this report, we refer to the impact of the 30% target on avoided-cost and non-monetary values as the **broad-sense economic** outcomes, to distinguish them from the **financial** outcomes.

Findings of the report

In the **multi-sector analysis of financial outcomes**, we found that total economic output is greater if the 30% target is implemented, than if it is not implemented. The projected increase in global output depends on the implementation approach taken, ranging from \$64 - \$454 billion per year by 2050 in our illustrative scenarios. These figures only represent the increase in direct expenditures (revenues) and do not include multiplier effects. Recent estimates of average multipliers range from 1.36 in agriculture to 3.2 in tourism^{20,22,24}, and so the final boost to global economic output may be over one trillion US dollars per year. This figure does not include novel revenue sources, such as green investments, biodiversity and climate bonds, and increased payments for ecosystem services^{25,26}. Although difficult to model post-2030, many such novel revenues are likely to arise if 30% of the planet belongs to a single underexploited type of asset (protected nature). The potential is illustrated by the rapid growth in “green” financial markets – for example, global Green Bond issuance has risen from \$11 billion to \$200 billion in only six years^{25,26}. The final benefits may therefore be much greater.

In the **PA/Nature sector**, much of the projected revenue growth comes from an increase in nature tourism and its associated benefits. Given recent travel restrictions associated with COVID-19, we emphasise that even if post-2030 nature tourism revenues were reduced to half their projected value, expanding PAs would still lead to a net output gain. Nevertheless, financial hedges or safety nets against sudden revenue shocks to the PA/nature sector might be needed, similar to the safety nets that exist for multiple other economic sectors.

Our models project that implementing the 30% target would also lead to increased output values (revenues) in **agriculture** (all crops and ruminant meat) and **forestry**. This effect is commonly observed when land-use is restricted, because scarcity of available land (or sea) pushes up the prices that producers are paid for their goods and also motivates productivity improvement²⁷⁻²⁹. We note that higher farm gate prices may imply a benefit to rural producers but a disbenefit to consumers (who will predominantly live in wealthier cities by 2050³⁰, accompanied by an approximate doubling of GDP per capita³¹). To gauge the social implications of the consumer price impact, it is important to note that food prices are projected to fall by 2050 and so in reality, the impact is “a slightly smaller reduction in prices than would otherwise be expected”. Even so, it would be important to monitor carefully any potential impact on the poor and especially, on the urban poor.

The analysis of producer revenues does not account for the possibility that PA expansion may also increase producer expenditures, due to intensification, limited land availability and higher land prices, or longer distance to markets. We were able to test for this in two agricultural models and found that overall, projections for changes in agricultural value added (producer revenues minus costs) ranged from small increases to small decreases, with an average of -0.49% (Figure 2B). By far the highest reduction (in both models) was for the GDN scenario (1.2% and 2.7%), which was due to increased expenditure on inputs and higher land prices. These worst-case values represent, at maximum, a few tens of billions of dollars globally, which is an order of magnitude smaller than the potential gains in the PA/nature sector if the 30% target is implemented. We comment that if producers have higher expenditures as well as higher revenues, the first beneficiaries of the higher expenditures are often the suppliers of agricultural inputs (e.g. fertilizers or labour). Further research would be needed to determine the impact on the agricultural sector as a whole.

For the **forestry sector**, implementing the 30% PA target again increased output values (revenues), driven by increases in efficiency and the price paid to producers when the availability of exploitable tree-covered land was reduced. Total roundwood output value reached \$428 billion in 2050 under the no-PA-expansion baseline, \$450 billion in the production-focused Three Conditions scenario, and over

\$500 billion (~20% higher than the baseline) for the three more biodiversity-focused scenarios (Save Species from Extinction, Biodiversity/Wilderness Consensus, and Global Deal for Nature). The forestry model does not generate future cost estimates, so net output effects were not analysable.

In the wild-capture **fisheries sector**, catch values are generally expected to decline post-2020 due to overfishing and climate change. We found that creating more marine PAs also causes an initial catch-value shock (decline), but then begins to have the reverse effect, increasing catch, catch per unit effort and revenue (Figure 4), as also observed in smaller-scale empirical studies³². Marine PAs can therefore act as a type of fisheries management, similar to existing no-take practices which place areas off limits to allow the natural resource to regenerate. However, if one considers the ocean economy as a whole, marine PAs have often been found to generate better local economic returns than fisheries, due to their attraction for divers, recreational boating, tourists, and other aspects of the marine leisure-and-recreation economy.^{32,33} In general, therefore, the ocean economy as a whole would benefit in all years, whereas wild-capture fisheries would initially suffer an income shock followed by a mid-term gain (in comparison to the business-as-usual expectation of an ongoing decline). We caution that this initial shock may last for several years – making it similar to a “loss leader” investment for a longer-term benefit. The impact on aquaculture was not analysable with current information (see Full Report).

For major, future-facing policy decisions that imply a large initial investment, it is also important to quantify how the policy is expected to change **future economic/GDP growth rates**. The modelling results suggest that the PA/nature sector's revenue is likely to grow at least seven times faster than that from agriculture and fisheries (5%-6% versus <1% for agriculture and a contraction for fisheries) (Figure 3). The growth of PA/nature revenues may be even higher than this, because if 30% of the planet prioritizes nature, then multiple innovative markets will develop means to extract revenue from it, also creating new employment. Implementing the 30% target can therefore be expected to lead to higher revenue growth overall, especially in rural and coastal economies.

In our **broad-sense economic analysis** (limited to two biomes in tropical countries only), we found that implementing the proposed 30% target would generate an additional economic benefit of \$170–\$534 billion per year by 2050, over and above the financial benefit. These values reflect the way that PAs prevent the conversion of natural structures that are critical for defence against floods and storm surges, reduction in carbon emissions that lead to climate change, and (an incomplete list of) other services^{14,34}. The total global economic value of PAs, including all countries and all ecosystem services, would clearly be much greater than this partial estimate. Protecting natural areas indeed reduces the risk of new zoonotic disease outbreaks such as COVID-19, a value that has not yet been quantified despite the very high economic costs of the recent pandemic^{35,36}.

Implementing the 30% target could also have benefits that would be difficult or inappropriate to quantify in monetary terms. Most obviously, the 30% target is designed to reduce the rate of biodiversity loss, a major component of international commitments to the CBD and to the Sustainable Development Goals^{13,37}. And if OECMs are adopted as a way to recognise the nature stewardship contribution of Indigenous peoples and local communities (hereinafter IPLC)^{19,38}, then our scenarios suggest that the amount of IPLC land recognised in this way could increase by 63-98% under the proposal.

The **investment needed** to implement the 30% target through protected areas ranges from \$103 billion per year to \$178 billion² per year depending on the implementation scenarios chosen on land and sea.

2 This range does not include the Global Deal for Nature, where the cost may be higher but cannot easily be modelled, due to a strong component of protected landscapes rather than protected areas in the traditional sense. Other scenarios, including the Biodiversity/Wilderness Consensus, may also be cheaper than modelled for the same reason.

This cost is divided up into the minimum budget needed to adequately manage the existing PA system (\$67.6 billion per year) and the cost of adding new areas to reach the 30% target (between \$35.5 billion and \$110 billion per year), although it is likely that costs at the upper end have been overestimated³. Importantly, higher-cost scenarios generate higher revenues, suggesting that pure focus on cost is counterproductive.

Knowing the **costs and benefits for the PA sector alone** also allows us to perform a traditional single-sector analysis on whether the benefits exceed the costs of investment. We found that benefits exceed costs in the purely financial analysis (irrespective of scenario), and more so if the financial and economic benefits are combined (Figure 5). A business-style benefit-to-cost ratio is not appropriate because the investors do not collect the financial rewards in this case. The overall picture, however, is that all approaches yield a net benefit, but the size of that benefit depends on the level of ambition (or the degree of affordability).

Different degrees of affordability and different **distributions of costs and benefits** both have important financial and social implications. Countries with poorly developed nature/PA sectors may have insufficient resources or capacity to capture the economic benefits (and opportunities) of expanding PA networks and developing their associated revenue streams. And yet, about 70%-90% of the implementation cost for the 30% target will fall on low- and middle-income countries, far more than their domestic public-spending budgets could support. The policy implication is that such countries should receive aid and assistance to capitalise on nature-based opportunities, in which they have a competitive asset advantage. Without such assistance, existing global economic inequalities will simply expand to include the rapidly-growing PA/nature sector, and sustainable development would be disincentivised instead of promoted.

Aggregate country-scale benefits can also conceal local losses caused by the creation of new PAs. Such local effects can generate considerable social and economic problems, reducing the effectiveness of PAs themselves. A carefully designed mix of local analysis, compensation, community support, livelihood alternatives, education, governance and spatial planning would therefore be a critical component of any implementation strategy.

³ The ranges for both total costs and additional costs do not include an unusually high estimate for the Global Deal for Nature (Table 3), see comments above on how that scenario has a strong landscape element that probably cannot be robustly modelled from existing data on more traditional PA costs.

FULL REPORT

Introduction: The 30% Target for Protected Areas and OECMs

Protected areas (PAs) represent one of the main strategic options to mitigate the current biodiversity crisis (the impending Sixth Mass Extinction)³⁻⁵. However, there is broad consensus that not enough land and sea is yet protected to achieve international targets on ‘significantly reducing’ biodiversity decline, nor to protect habitats critical to maintaining a safe planetary operating space^{7,9-12,37,39}. Calls to increase biodiversity conservation areas come not only from conservation biologists, but also from the World Economic Forum, which ranks biodiversity loss as one of the top five threats to the global economy², and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES)^{1,40}. The flagship report of the UNEP (Global Environment Outlook 6th Assessment) also recognizes the cost of inaction on biodiversity conservation to be of considerable importance⁴¹. One of the major proposals for the 15th Conference of Parties of the Convention on Biological Diversity is therefore to increase global PA and OECM coverage to at least 30% between 2020 and 2030 (Target 2 in the draft Post 2020 Global Biodiversity Framework¹³). Currently, -16% of the land and 7.4% of the ocean is in areas designated or proposed for protection (and only 2.5% of the ocean is in highly/fully protected areas)^{7,8}.

The prospect of having 30% of all land and sea protected raises a number of economic concerns about potential negative impacts on economic output and local communities. A first concern is that agricultural or fisheries production may be compromised. There are also basic budget concerns about how much a very large PA system may cost. Biologists have almost the opposite concern: expanding an already-underfunded global PA system could simply lead to the creation of even more paper parks, hitting headline political targets on areas conserved but failing to achieve the underlying intention of saving biodiversity⁴².

However, these concerns all need to be weighed against the benefits of PAs. Traditionally, those benefits have been seen as non-monetary but today, PAs also generate considerable financial benefits. For example, nature tourism is a major contributor to global GDP, estimated to contribute between \$344 billion and \$600 billion per year^{43,44}, and the number of tourists to Africa is expected to double by 2030, with 80% of those visits driven by wildlife watching in the continent’s nature sector^{45,46} (although the current COVID-19-related fall in travel might now delay the doubling date to a few years later). Countries as diverse as Tanzania, Costa Rica, Ecuador, Palau and Australia are all aware of the enormous benefit to the economy that protected-area wildlife, scenic beauty and coral reefs can bring. A similar picture is seen across the Caribbean, Asia, the Americas, Oceania and the Pacific.

In many countries, there is already a clear economic case for investing in PAs, because they generate more economic benefit than they cost to run. In the United States, a study found that every \$1 invested in the National Park system returned \$10 to the economy (with the PA system also providing 295,000 jobs)⁴⁷. Thanks to the growth of revenue-generating opportunities in PAs from income flows as diverse as bioprospecting payments for antiviral, anticancer and other drugs, tourism, film credits, carbon credits, and payments for brand equity values, the modern view is that PAs (and nature conservation more broadly) may actually be financially beneficial to a national economy, rather than being non-productive

and outside the economy as historically thought. We emphasise that the increasing level of cash revenue generated by natural areas does not imply that PAs have a value as “merely as a commodity”. We only wish to explore whether there is now a revenue-based for greater investment in them, to complement the case based on their non-monetary value as public goods.

A revenue-based case is easily confused with a different economic argument: that nature provides many “ecosystem services” that have very large importance for human wellbeing (which therefore has value). Some of the largest ecosystem service values (particularly those for regulating services) relate to *avoided costs*, defined as the likely future cost of ignoring a major risk. Avoided-cost spending is widespread in all public budgets based on some form of defence against risk (including military defence, flood defence, law and order, and public health, as evidenced by recent government responses to the coronavirus epidemic). The basic principle is that investment in preventing or mitigating a potentially catastrophic risk represents sound economic policy strategy, even if risk avoidance itself does not necessarily generate revenues. And nature supplies many defences against outcomes that would otherwise be economically catastrophic. Mangroves can prevent ocean storm surges from causing millions of dollars’ worth of damage to coastal settlements and farm fields (although often, this value is only discovered after the mangroves have been cut down)^{14,48}. One third of the world’s largest cities depend on protected areas to maintain the cleanliness and supply of a significant portion of their drinking water²³. Removal of unprotected forests causes multiple economic losses from topsoil removal, flooding and water pollution, as well as driving climate change and its multiple economic risks^{27,34,49,50}. Human health also suffers in multiple ways from the absence of natural areas, from stunted growth in children⁵¹, to increased risk of zoonotic disease transmission³⁵, to multiple physical and mental health benefits documented for brief exposure to forests as an outlet for stress⁵².

It is for all these reasons that the World Economic Forum now ranks biodiversity loss as one of the top risks to the global economy. (For a fuller description and discussion of ecosystem service values, see the McKinnon et al. 2019³⁶ and the IPBES Global Assessment Report¹.) The costs of not investing in risk reduction, for example by allowing natural capital to degrade, are often discovered only after a catastrophe has occurred. But despite such warnings about avoided risks and ecosystem service values, such values seem to have influenced decision-making to a limited degree⁵³. The difficulty is that ecosystem service values often equate to “shadow values” that are either non-monetised (not revenue-generating) or incompletely monetised. Countries could typically spend much more on reducing risks but have not done so because more immediately concrete priorities have taken priority, such as day-to-day social or economic needs. Put simply, *financial* considerations (in the narrow sense of concrete cashflow) have often overcome *economic* considerations (in the broad sense of “economic” that includes both monetary and non-monetary values).

For that reason, we divide our analysis of the proposed 30% target into two parts. We first carry out a *financial* analysis of costs and benefits, focusing on what some might term “real money” flows. We then address the question of a wider *economic* analysis, considering the potential value of imperfectly monetised ecosystem services. We note that the broader economic analysis is necessarily more limited in scope because, in the short policy timeline given, it was not feasible to quantify how multiple PA expansion scenarios might change both the supply and demand of ecosystem services and their value at a global scale, compared to a counterfactual of non-PA-expansion. We therefore focus on some of the key ecosystem types for which it is possible to estimate a global: forests and mangroves⁴.

4 Coral reefs have also been widely valued but so much of that value is associated with tourism (which we already include in our financial analysis) that to avoid double counting, it is not included in the list of ecosystems for ecosystem service valuation. Moreover, much of the protection a coral reef offers against wave action seems unlikely to be affected by whether or not the reef is in a formally protected area or not, at least in our 2030-2050 focal period.

Purpose and Framework of this Report: Benefits and Costs

The main purpose of this report is to quantify the costs and benefits of the proposed 30% target, including opportunity costs where PAs compete with alternative (traditionally productive) land/ocean uses. The report is primarily designed for national-level and international decision-making, and thus largely focuses on the interests of the sovereign governments that are signatories to the CBD, and on international stakeholders in the fields of biodiversity, agriculture, forestry, fisheries, the Blue Economy⁵⁴ and the Sustainable Development Goals. The report does not focus on OECMs due to the fact that the term was defined by the CBD only in 2018.

We first carry out a standard appraisal (a *financial* analysis) of how policy decisions related to the 30% target might affect total economic output across the multiple sectors most directly impacted – agriculture, forestry, fisheries, tourism and the flow of nature-based revenues itself, which we refer to as the “PA/nature sector”. This appraisal framework recognises that any land-use or ocean-use policy decision will favour some sectors/activities and restrict others. Each sector will communicate (and lobby for) the benefits if it is selected, and the opportunity costs if it is not. But at the highest decision-making level, the relevant authorities will consider which decision is likely to generate the greatest benefit for the whole economy (or more widely, for the whole of society)⁵⁵⁻⁶⁰. In this report, we apply that classic economic decision framework, summing up economic outputs across the main sectors that would compete with PAs in area-use decisions, and asking which option generates the greatest total economic output when all the sectors are considered together.

On land, most new PAs are likely to be declared on natural or semi-natural land, where the main alternative uses (competing sectors) are agriculture and forestry. Marine protected areas are most likely to compete with fisheries for ocean use, making fisheries our third sector for analysis. We then define a fourth sector, the “PA/nature sector” itself, to emphasise that PAs are very much part of a country’s economy and can drive considerable economic activity (for example from expenditures by nature tourists and visitors, which have an estimated value of \$600 billion per year⁴⁴). The total benefit in the multi-sector analysis is therefore defined as the sum of all the relevant economic outputs for all the competing sectors, recognising that each policy option may represent higher or lower outputs for individual sectors, but taking the total benefit as the assumed high-level goal. The proposed 30% target is for implementation by 2030 and would be followed by a period in which other land-use and ocean-use sectors adjust their pattern of expansion and distribution in response. We therefore focused on estimating economic outcomes that lay in the near future beyond the 2030 target date, taking 2050 (the goal date of the CBD Mission and Sustainable Development Agenda³⁷) as our main reporting year, with an interim calculation for 2040.

Immediately, we emphasise that monetary benefit is not the only driver of government decisions or societal value. Although financial questions are central to policy making, non-monetary or social, non-revenue-generating considerations are also important. Ideally, however, policies might be hoped to fulfil multiple roles – generating revenue, improving social and human wellbeing, and reducing economic risks simultaneously.

Alongside the first concern that expanding PAs may act as a drag on the economy, a second concern is that the budget for a larger and fully-funded system may be very high – with the main burden expected to fall on government budgets or international aid donors. In this report, we therefore examine what the costs of the 30% target might be.

Since PAs themselves have both costs and benefits, we also conduct a single-sector analysis that compared the costs of an expanded PA system to the financial benefits. If the benefits are larger than costs, investing in PAs would be more akin pro-economy public-spending investments such as road

building, changing the historical narrative that public spending on PAs supports a public good but contributes little to the economy.

We recognise that the COVID-19 pandemic may currently be limiting public spending and investment capacity, but the 30% target would be implemented in future years, precisely at a point where stimulus investment to re-grow the global economy will be needed. We also note that this report necessarily focuses on PAs such as National Parks or marine parks because these have a history long enough to provide data on likely costs. Costs of OECMs remain largely unknown and would need to be explored on a case by case basis. However, we assume that PAs are subject to the same variety of use restrictions (and permissions) already in place, including a component where sustainable logging or other sustainable resource uses may be allowed.

Creating Different Scenarios for how the 30% Target might be Implemented

Sovereign governments must ultimately decide for themselves where any new protected areas should be positioned on land and sea, with the likely decision outcomes still unknown. Nevertheless, for economic analysis, it was necessary to generate a set of options about where the new PAs might be located to achieve a 30% target, not least because PA location (e.g. in remote areas or close to human activity) could greatly affect the ultimate benefits and costs that arise. There may also be value in knowing the opinions of biologists and economists on where new protection might be most appropriate.

We therefore created six terrestrial and five marine scenarios of how the 30% target might be implemented in spatial terms, plus an additional baseline scenario (called the “no-PA-expansion” scenario). Decisions about appropriate PA location depend on a wide variety of views about what values PAs should protect (“biological need”), and about how much those biological needs should compromise with other industries or social needs. For biological needs, we canvassed a broad array of global biodiversity experts and through them, sourced a set of global maps (“scenarios”) for a 30% PA system, all of them based heavily on biodiversity priorities. We then considered how the trade-off between ideal biological need and political and economic realities might change the positioning of PAs, developing further maps that reflected an appreciation of those trade-offs. All scenarios therefore fit into one of three broad categories: (1) scenarios that focus more strongly on biological priorities, referred to as “biodiversity-focused” PA scenarios; (2) scenarios that prioritize production (placing PAs in biologically important places that nevertheless avoid conflict with agricultural and fisheries production), referred to as “production-focused” scenarios; (3) a halfway compromise between biodiversity and production priorities, referred to as “biodiversity/production compromise” scenarios (Table 1). Due to disciplinary specialisation, terrestrial and marine PA-expansion scenarios were generated separately and so finally, they were paired together to generate six overarching scenarios protecting both the terrestrial and marine realms to a coverage total of 30% (Table 2).

TABLE 1.**Terrestrial and marine protected-area scenarios used to explore the economic implications of the 30% target.**

Note that there is no M2 or M7 scenario because the terrestrial scenarios T2 & T7 (Three Conditions and Global Deal for Nature) did not have an exact marine equivalent. T = terrestrial realm, M = marine realm; MPA = marine protected area.

Realm	ID	Category	Name	Description
T	T1	Non-PA-expansion baseline	Reference Scenario (REF)	The counterfactual for comparison of not expanding protected areas post-2020. Maintain the protected area estate at its current coverage (as of July 2019).
M	M1	Non-PA-expansion baseline	Reference Scenario (REF)	The counterfactual for comparison of not expanding protected areas post-2020. Maintain the marine protected area estate at its current coverage (as of July 2019).
T	T2	Production-focused	Three Conditions (THC)	Protect all global habitat that retains a state of minimal human intervention ('wilderness'), plus existing PAs
T	T3	Production-focused	Harsh Political Reality (HPR)	First disqualify from protection any area needed for agricultural production up to 2050, then choose the areas of non-disqualified land that optimally minimises global species extinctions (plus existing PAs).
M	M3	Production-focused	Harsh Political Reality Marine (HPR)	Marine reserves are not allowed on any areas of ocean that are currently high-value fishing grounds. The existing MPA system is then expanded to the next-best set of ocean sites to preserve marine biodiversity
T	T4	Biodiversity-focused (hybrid)	Biodiversity/Wilderness consensus (BIWI)	Protect all wilderness, KBAs and existing PAs, plus the optimal set of all other sites needed to maintain global species viabilities (based on minimum range coverage). NB 43% terrestrial coverage, compensated by lower coverage in the paired marine scenario
M	M4	Biodiversity-focused (hybrid)	Biodiversity/Wilderness consensus (BIWI)	All existing MPAs and marine wildernesses are protected, then the optimal set of areas needed to prevent marine biodiversity decline (NB cover 26% of marine global area)
T	T5	Biodiversity/Production Compromise (BPC)	Biodiversity/Production Compromise	Add -5% more of the land surface to the existing PA network (up to 20% planetary land), choosing sites to optimally reduce global species extinctions. The remaining 10% of new PAs are not allowed to go on potential agricultural land, but are placed in the next-best set of sites instead (using the same criteria)
M	M5a	Biodiversity/Production Compromise	50:50 EEZ (5050EEZ)	Expand the existing MPA system to the 30% of ocean that optimally reduce global species extinctions, but 50% of the protected area inside Exclusive Economic Zones permits sustainable fishing, while the other 50% bans all economic activity
M	M5b	Biodiversity/Production Compromise	50:50 Coastal (5050COAST)	Expand the existing MPA system to the 30% of ocean that optimally reduce global species extinctions, but 50% of the protected area immediately adjacent to coasts (where small scale fisheries tend to operate) permits sustainable fishing, while the other 50% bans all economic activity
T	T6	Biodiversity-focused	Save Species from Extinction (SSE)	Expand the existing PA system to the 30% of land that optimally reduce global species extinctions
T	T7	Biodiversity-focused	Global Deal for Nature (GDN)	Conserve a wide range of sites that have biodiversity importance under different criteria, including current PAs, sites with rare or endemic species, areas needed for wide-ranging mammals, etc. (see original paper).
M	M6	Biodiversity-focused	Top 30 (TOP30)	Expand the existing MPA system to the 30% of ocean that optimally reduce global species extinctions

TABLE 2.**Scenarios for the full 30% target, including both terrestrial and marine protected areas.**

Terrestrial and marine scenarios were paired based on their broad category and on more detailed similarities of approach. Note that Marine Harsh Political Reality and Top30 are both paired twice because in the terrestrial realm, Three Conditions and the Global Deal for Nature had no exact marine equivalent, and so we duplicated the closest marine equivalents to create the necessary pairs. There were two possible marine pairings for the terrestrial Biodiversity/Production Compromise scenario (50:50 EEZ and 50:50 Coastal) but for this report, we used 50:50 EEZ.

Paired scenario	Terrestrial component	Marine component	Category
REF	Reference scenario	Reference scenario	Non-PA-expansion (baseline)
THC	Three Conditions	Harsh Political Reality (marine)	Production-focused
HPR	Harsh Political Reality	Harsh Political Reality (marine)	Production-focused
BPC	Biodiversity/Production Compromise	50:50 EEZ	Biodiversity/Production Compromise
BIWI	Biodiversity/wilderness consensus	Biodiversity/wilderness consensus (marine)	Biodiversity-focused
SSE	Save Species from Extinction	Top 30	Biodiversity-focused
GDN	Global Deal for Nature	Top 30	Biodiversity-focused

Estimating Total and Sector-specific Economic Outputs for the Different Policy Options

As stated above, the main economic sectors that might have their activities redistributed or constrained by an expansion in the PA system are agriculture, forestry and fisheries. To these should be added the PA/nature sector, which can also generate revenues (e.g. through nature tourism) that sometimes exceed the income generated by other productive sectors. We therefore quantified the projected changes in output across all those sectors, under each of the 30% scenarios (including the no-PA-expansion scenario, for comparison). We did not attempt to analyse how new protected areas might cause a shift in the pattern of post-2030 fossil energy and mineral exploration and extraction, because such information would be very hard to predict. Nor was there sufficient information available on the global distribution of aquaculture operations and where they were likely to redistribute to post-2030 in response of a putative MPA expansion (although we note that often, aquaculture may be expected to increase its operations to compensate for the impact of a very large MPA expansion on wild-capture fisheries). Finally, PAs may potentially affect future transportation routes, but the cost and revenue implications would be extremely complex to model, and were considered beyond the scope of this study.

Methods (Financial Analysis)

PA/NATURE SECTOR OUTPUTS/REVENUES

For the PA/nature sector, which is only recently beginning to realise its revenue-generating potential, it is important to distinguish between current revenue sources and potential future revenue sources. The main current revenue source is nature tourism, which represents an increasingly important subsector of global travel and tourism. Over the past few decades, the global tourism sector has grown faster than GDP, despite a major shock during the 2008/2009 global financial crisis (which looks likely to be repeated during the coronavirus pandemic), and nature tourism has showed a similar pattern.^{43,61} To estimate the likely post-2030 revenues from an expanded global PA system, we collated global information on current patterns in visitor numbers and expenditures for national protected area networks, then used predictive statistical models and a time-series analysis of trends in 13 years of social media postings from protected area visits to forecast post-2030 visitor revenues under different scenarios. In doing so, we did not assume naively that tourism would double if protected areas double. Instead, we projected forward from the existing pattern of visitor increases (which include a major dip during the 2009 financial crisis), and then used the empirically observed effects of greater accessibility and market signalling on visitor numbers to estimate the additional visitors expected when more protected areas are created. We checked our nature tourism growth analyses against the known patterns of recent visitor increase, confirming that our models concurred closely with the approximately 4% growth currently observed (but more if there was a sudden, large increase in the number of accessible protected areas). Our social media analysis also accounted for differences in the number of social media users in each country and each year.

Another possible source of revenue is a future global carbon market, to address the climate change crisis⁶². However, it is extremely difficult to predict what the market price of carbon might be in 2050 and 2040, or what reference baselines will be required. We therefore assess avoided-emissions values in the broad-sense economic analysis below, rather than in the financial analysis here.

A third potential revenue source post-2030 is private and corporate enterprises⁶³, which currently make little contribution to PAs. Potential private and corporate funding to biodiversity has already been estimated at \$50 billion, for present-day conservation arrangements⁶³. However, post-2030, this revenue capacity would be likely to increase if 30% of the earth became protected, as donations, corporate social responsibility, and finance-industry innovations would have a greater stock of formalised natural capital to work with. Based on the \$50 billion figure already suggested (that considered only the existing protected area system), we used \$50-\$60 billion as an estimate of the private revenue-generating potential of a nature economy with 30% PAs (depending on the level of nature protection in each scenario). These values may strongly underestimate the true private-sector revenue generated by a 30% PA system (revenues which could also include offsetting income^{64,65}), but there are counter-opinions that private corporations are unlikely to assist in protecting the public good of global biodiversity, and so we preferred a potentially conservative estimate.

Aid, as a direct cashflow consequence of the existence of a sector, formally contributes to GDP. Most aid data for conservation refers to biodiversity aid, and we assumed that aid could double by 2050 (to 1/10,000th of global GDP) as a response to the expansion of conservation areas. Not all current biodiversity aid is associated with protected areas, but a 30% target would encompass a large fraction of the areas needed for biodiversity conservation, implying an increase in PA-associated flows. However, possible errors in this estimate are unlikely to affect our broader conclusions because the aid component is only about 2-3% of the total revenues from PAs. Aid is included as a gross flow but not a net flow, and is more relevant at the country scale than at the global scale, because it might be perceived as a cost by the donor, but as a revenue by the recipient.

AGRICULTURAL SECTOR OUTPUTS/REVENUES

On land, the main effect of creating new protected areas is to regulate where new agricultural areas can be placed. If farm expansion occurs into a slightly more expensive or less productive area, then the opportunity cost is the income difference between the first-choice and second-choice areas. The instinctive fear that ‘every hectare of forest protected represents a lost hectare of agricultural land’ is rarely true, because a second-choice area for conversion to a new field is often available elsewhere in the country (although importantly, local opportunity costs are still possible and different from national-scale opportunity costs). The key question, therefore, is whether the adjusted expansion of agriculture, triggered by the implementation of 30% PAs, causes an important difference in revenues *as compared to the no-PA-expansion scenario*. At the national level of decision making, moving agricultural expansion to the second-choice area may still be the preferred policy if an alternative use of the first-choice area generates greater economic (or social) benefits.

To calculate projected revenues for the agricultural sector when the 30% target is implemented, we used FAOSTAT data to estimate the 2010 revenue at global scale for all crops and for ruminant meat (2,872 billion 2015 USD), and then used the land-use modules of four Integrated Assessment Models (IAMs) to project how agriculture could develop under the different scenarios: AIM, MAGPIE, IMAGE and GLOBIOM (see Popp et al, and Stehfest et al, for overviews^{66,67}). These global land use modelling systems take assumptions on future trends in human population growth, economic growth, food preferences, technological development and global trade patterns, to project the future of agricultural demand, production and trade with a time step of 5 to 10 years. They also project at the scale of 0.5 degree (around 50 km at the equator) or finer the associated changes in the extent of agricultural land, based on current land-use, datasets on soil fertility and costs, and economic theory. They also output at regional level the area, productivity and farmgate prices projected for the entire agricultural sector. The baseline scenario was based on the SSP2 Shared Socioeconomic Pathway, picturing a world in which historical trends are continuing into the future (see Popp et al. 2017⁶⁶). The various scenarios were introduced as restrictions on land use change (protected areas were not allowed to be converted), and the models were used to derive the resulting changes in agricultural productivity and extent, and in the price paid for agricultural products. (We note that these projections did not incorporate climate forcing effects.) The use of four different land-use models however allows accounting for uncertainties associated with future projections and PA effects. An estimate of the future revenue projected by each land-use model and scenario for the years 2010 to 2050 was generated by multiplying the present-day revenue estimated from FAOSTAT data by the model-, time- and scenario-specific indices of agricultural area, productivity and price. To identify robust trends and uncertainties in outcomes associated to the different scenarios, we calculated the mean and range across the four models.

Although production is by far the largest revenue source in agriculture, for consistency, we included international agricultural aid as a source of agriculture-sector revenue, as we had for the other sectors. Agricultural aid, however, represented a very small proportion (<1%) of total projected agricultural revenues.

In addition to calculating gross output values, we were also able to estimate changes in agricultural value added (revenues minus costs) for two of the four models (AIM, IMAGE).

FORESTRY SECTOR OUTPUTS/REVENUES

To calculate forestry sector revenues, we inserted the six terrestrial scenarios (plus the reference baseline scenario) into the Global Forest Products Model (GFPM)⁶⁸⁻⁷⁰, applying the same implementation timetable as agriculture in the IAMs. The GFPM calculates changes in forestry activity, trade and prices

and requires, as input, estimates of the area of exploitable and non-exploitable forest in each of 180 countries⁶⁸⁻⁷⁰. It then outputs the total (gross) value of forestry products from the exploitable portion, and the carbon stock in all forests (protected and unprotected), for ten-year time intervals. The calculations include estimates of forest growth (and hence, carbon sequestration) and, for the purposes of the current calculation, the model was extended to calculate the changing carbon stocks in non-exploitable (protected) forest as well as exploitable forest. Initial exploitable forest cover was calculated by taking the current protected area maps for each country and overlaying them on the 2015 ESA CCI land use gridded product⁷¹, interpreting land use classes 50-90 as “forest”. Exploitable and non-exploitable (protected) forest cover for the scenarios was calculated similarly. Some classes of protected areas (especially IUCN categories V and VI) can allow sustainable use of resources, including wood from forests. We therefore calculated the proportion of category V and VI protected areas in each country currently and assumed a similar proportion in the post-2020 scenarios, adjusting the exploitable forest area accordingly.

FISHERIES SECTOR OUTPUTS/REVENUES AND CATCH

Protected areas affect fisheries in a notably different way from agriculture because they can frequently result in closure of areas that are currently being exploited. However, several theoretical models and case studies suggest that protection helps fish stocks inside the marine reserve to recover, but over timescales ranging from a few years to decades, and probably depending on the level of pre-protection degradation and the characteristics of the local ecosystem⁷². Many scientists have also hypothesized that because fish are mobile organisms, the recovery inside the reserve should lead to “spillover” – an increase in the fish stocks available to fishermen outside the reserve due to stock recoveries occurring inside the reserve. MPAs would therefore provide a commercial benefit similar to many traditional management practices, in which fishers themselves leave some areas unexploited to allow stock recovery^{73,74}. In layman’s terms, this can create a situation of “short-term pain for long-term gain”. There is evidence of this spillover benefit in observational studies, although the timescales are less well studied^{75,76}.

We estimated how implementation of the 30% target would affect post-2030 fisheries revenue using three “Marine Ecosystem Models” (MEMs)⁷⁷: BOATS^{78,79}, EcoOcean⁸⁰⁻⁸² and DBEM^{83,84}. These models generate projections for changes in species abundance and distributions, fisheries catch, efficiency and cost, with and without implementation of 30% MPAs, in each country. We define a country as the Exclusive Economic Zone (EEZ) belonging to that country, recognising that some territorial uncertainties exist and that EEZ biomass is sometimes captured by other countries. Given current progress on climate agreements (a likely track towards greater than 2 degrees Celsius warming⁸⁵), the ocean models took a path midway between optimism and extreme pessimism, using the predicted outcomes under two different levels of climate forcing: RCP4.5 and RCP6.0, representing a low-medium level of climate change and a medium-high level. Up to 2050, the outcomes of these two climate change predictions were very similar, so we report the RCP6.0 scenario in the main results.

As for other sectors, we added fisheries aid to the revenues generated by the presence of the industry in developing countries, projecting current aid trends forward to 2050 (see Methods Appendix). We note that fisheries aid represents less than 1% of all fisheries-related revenue flows.

Results (Financial Analysis)

FINANCIAL BENEFITS OF IMPLEMENTING THE 30% TARGET

In the financial multi-sector analysis, projected 2050 revenues from all four sectors combined were between \$64 billion and \$454 billion higher than under the no-PA-expansion alternative (depending on scenario, Figure 1, Table 3). These estimates of benefit are highly conservative because they ignore the possibility of innovation in revenue capture when 30% of the planetary asset base (by area) is protected. Indeed, future carbon-market revenues could generate considerably more income to the PA/nature sector.

We also conducted an analysis to see whether benefits outweighed costs in the PA/nature sector alone (which would also be a form of justification for investment in the 30% target). The benefits consistently outweighed the costs, whether we used direct visitor spending values only; or PA revenues boosted by a multiplier to reflect the benefits to supply chains and downstream economic activity; or the combination of (direct) financial values plus broader-sense economic values.

BREAKDOWN OF OUTPUTS/REVENUES AND OPPORTUNITY COSTS BY SECTOR

PA/Nature

For the PA/nature sector, the main flows projected in 2050 were from recreation and nature tourism (\$330 billion to \$484 billion in direct tourist expenditure). Importantly, expanding to 30% PAs outperformed the no-PA-expansion option even when we artificially tested halving tourism revenues, because a globally reduced tourism industry affected both the no-PA-expansion revenues and the 30%-PA revenues. The highest combined revenues (and the highest net multi-sector benefits) were projected in scenarios that place protected areas where visitors can access them easily. In our post-2020 calculations, the traditional practice of relegating protected areas to places that are remote or have low agricultural potential^{86,87} had the poorest financial performance because gains to production industries (avoided opportunity costs) were relatively small when agricultural expansion was redistributed, but losses to the PA/nature sector were relatively large when nature reserves were remote and inaccessible. We do not mean to suggest that we should not conserve wildernesses such as the Amazon or the arctic tundra. Remote wilderness areas have multiple values for the maintenance of biodiversity and earth systems^{88,89}. A hybrid strategy of giving traditional styles of protection to threatened natural areas close to areas of major human use, while also classing non-threatened wildernesses as OECMs with long-term restrictions on conversion or exploitation, may be appropriate. Smaller flows came from biodiversity aid (which, even when doubled, was still projected to be only 1/10,000th of global GDP) and from other private sources.

Agriculture

In our models, global agriculture-sector output/revenue was predicted to increase modestly if the 30% target is implemented. Depending on the scenario, the increase expected was \$35 - \$181 billion. Such revenue increases are commonly observed when land-use is restricted, because the resulting scarcity of available land pushes up the prices that producers are paid for their goods and also motivates productivity improvement^{27-29,67}. In the model outputs, productivity indeed went up by an average +0.9% compared to the no-PA-expansion baseline (range -0.3% to +4.9%) for cropland (crops other than second-generation bioenergy), and +0.9% (range 0% to +3.9%) for grassland. The price patterns were also as theoretically expected: although the models project a decrease in the prices of agricultural commodities irrespective of scenario, prices are “higher” (not as reduced) in the 30% scenarios by an

average of 2.6% for crops (range of -0.1% to 12.6%) and 1.6% for more ruminant meat (range -0.1% to 6.3%). We caution that this price effect should not be overinterpreted because future developments of agricultural prices remain uncertain⁹⁰ and the price change implied the price change in fact is “a slightly smaller reduction in consumer prices than would otherwise be expected”. Furthermore, GDP per capita is expected to increase substantially (and by far more than the price change) over the same period⁵¹. However, effects may be more heterogeneous on the regional and local scale and need further research.

The analysis of producer revenues does not account for the possibility that PA expansion may also increase producer expenditures, due for example to intensification, higher land prices or longer distance to markets. An estimate of change in agricultural value added (revenues minus costs) was possible for two of the four models (Figure 2B). While one model shows a range of possible outcomes from a small increase to a small decrease in net output, the other model indicates possible losses in agricultural value added by up to 1.2 %, due to limited land availability and increase in land prices. However, the absolute value of such a loss would be a few tens of billions of dollars globally and in the broader multi-sector framework, this would be more than compensated for by the net financial gains in the PA/nature sector, which are an order of magnitude larger.

Forestry

For the forestry sector, there was a similar pattern of increase in output values (revenues), again driven by an increase in efficiency and price when the availability of exploitable tree-covered land was reduced by nature protection. Total roundwood output value reached \$428 billion in 2050 under the no-PA-expansion baseline, \$450 billion in the production-focused Three Conditions scenario, and over \$500 billion (~20% higher than the baseline) for the three biodiversity-focused scenarios. The forestry model does not generate future cost estimates, so net output effects were not analysable.

Fisheries

Fisheries revenues depended more strongly on which scenario was implemented. It is important to note first that even with no constraint on fishing (the non-PA-expansion scenario), fisheries incomes are expected to continue the pattern of decline already observed in recent years, driven by overfishing and likely to be exacerbated by climate change (as confirmed by our model projections for the non-PA-expansion scenario, Figure 4).

Interestingly, the no-PA-expansion scenario and the Biodiversity/Production Compromise scenario (50:50EEZ) both showed a brief reversal in decline between 2030 and 2040 (Figure 4), which seems likely to be a consequence of the assumption in the model that the entire MPA network present in 2020 should move from 2.5% strict no-take areas, to either all 7.4% or current MPAs being strict no-take (for the no-PA-expansion scenario), or having an existing 7.4% strict no-take area plus an additional 22.6% area that mandated a mixture sustainable fishing and no-take areas (the 50:50EEZ scenario). The implication is that increasing the strictness of protection would lead to spillover benefits where overexploited stocks started to recover, and that implementing a compromise protection would augment this effect. However, the effect was not sustainable over time, with wild-capture revenues in both of those scenarios then declining again after 2040.

The scenarios where 30% of the ocean was immediately closed to fishing (100% no-take MPAs) showed the opposite pattern, with revenues declining between 2030 and 2040, but rapid recovering after 2040 (at the same moment when the no-PA-expansion option returned to a decline) (Figure 4). The EcoOcean model, which focused on catch biomass rather than catch value, further suggested that the rate at which stocks recovered was directly related to the size of the initial drop in biomass. For example,

for 50:50EEZ, this model projected a 10% short-term decrease in catch (compared to the no-expansion baseline) but no clear subsequent improvement (Figure 4). In contrast, the Top30 scenario (which is 30% strict no-take) had a large short-term drop in revenue, but this was followed by an increase in catch over nine times larger than the no-PA-expansion scenario (Figure 4).

If focusing on the fisheries sector only, these patterns suggest that early compromise solutions, in which existing MPAs are respected as no-take zones and new MPAs allow some sustainable fishing, could generate short-term benefits and buffer the shock of what would otherwise be a major transition (e.g. from 2.5% strict protection to potentially, 30% strict protection). However, the modelling also emphasises that the full economic benefits to fisheries will only materialise if compromises are short-term and are progressively phased out over e.g. a decade, in favour of stricter protection (if it were deemed worthwhile at all to sacrifice speed of recovery to short-term needs).

If focusing on the broader ocean economy, it is noteworthy that tourism revenue in a 30% MPA system would be at least \$197 billion per year. This value greatly exceeds the \$6-9 billion short-term loss caused to the fisheries sector by initial MPA expansion. A pattern of marine tourism replacing declining fisheries confirms the results of several case studies in which the tourism gains from MPAs outweigh the short-term losses to local fisheries, and in which the fisheries themselves observed a subsequent increase in their catch, despite the initially-concerning constraint on permitted fishing grounds³².

Even so, an initial decline in catch, followed by a fisheries recovery, would suggest a need for policy measures to support traditional fishing communities as they recover towards greater commercial sustainability, similar to the ones implemented for commercial closures. For example, when cod fisheries in Eastern Canada were closed due to a collapse from overfishing in the 1990s, the immediate annual loss was the \$134 million value of the cod catch, but a five-year government program of \$1.9 billion was implemented as compensatory support⁹¹.

In summary, the models suggest that for fisheries, the impact of implementing a 30% MPA target could be initially negative (the first shock of closures), but would become positive later as stocks recovered. Without MPA expansion, fisheries catches are projected to suffer a decline in future, due to overexploitation exacerbated by climate change^{83,92}. Such a decline could have severe consequences for food security and protein intake, depending on whether aquaculture can replace the lost catch⁹³. In the marine economy as a whole, revenues from tourism in a 30% MPA system would be considerably higher than fisheries losses, leading to a net positive outcome, but with consequences for fishing livelihoods that would need to be addressed and compensated.

REVENUE GROWTH IN THE NATURE SECTOR, COMPARED TO OTHER SECTORS AND OVERALL GDP

Economic growth is often one of the primary measures of macroeconomic health. We therefore calculated how revenues were expected to grow for 2030-2050 under the different policy options. In our results, the nature sector was the fastest-growing of the competing industries, growing its revenues at 5-6% per year, compared less than 1% for agriculture and a contraction (revenue decline) in fisheries. Implementing the 30% target increased growth in the PA/nature sector itself, and also increased the rate of growth in the combined revenues of all the sectors considered, with biodiversity-focused implementation options having the highest growth rate (Figure 3).

In addition to its importance for the economy, fast economic growth is also important when a large public investment may be needed, especially if the investment causes the economy and tax revenues to grow faster than the interest on borrowing. The PA/nature sector's rapid growth potential therefore suggests a viable long-term investment that supports the economy more broadly. Indeed, the multiplier

effect of tourism is greater than the (already-strong) multiplier effect of agriculture²⁰⁻²², and so each percentage point of growth in direct revenues in the PA/nature sector drives even greater growth in the economy as a whole, and thus in the tax revenues returning to the public purse.

Broad-sense Economic and Non-monetary Values

Financial revenue flows and GDP are only one part of the social priorities that governments and public spending need to support. Governments and societies also need to consider non-monetary values, including social needs, and also defensive spending (where the focus is on mitigating the risk of catastrophic economic and social outcomes, rather than generating revenue *per se*, with recent spending on the coronavirus outbreak providing a clear example). The more commonly-understood types of defensive spending are military defence, flood defence and public health, all of which reduce the probability (risk) of economically and socially disastrous future events (e.g. war, flooding and epidemics). The recent economic warnings from the World Economic Forum and IPBES^{1,2} – that biodiversity loss represents one of the greatest current risks to the world economy – are essentially a call for defensive spending to avoid catastrophic risks associated with the continued loss of nature. Examples are the multi-million dollar damage to coastal infrastructure when mangroves and coastal wetlands are removed (and can no longer buffer storm surges); increased flooding and loss of agricultural soil if hillside forests are removed; and the global economic impacts of climate change if natural vegetation is cleared, releasing its carbon into the atmosphere. These avoided costs are just one aspect of the ecosystem service value of natural areas (all of which would ideally be quantified), but they can be some of the most important economically.

Our analysis was not able to carry out a full global analysis of the avoided-cost or ecosystem service values that might arise under multiple different global PA scenarios between 2030 and 2050. We therefore focused on a small subset of ecosystem service values that was analytically tractable in the time available for this report – those associated with conserving tropical forests and coastal mangroves, plus the value of forest carbon associated with avoided deforestation.

An important point to note is that we did not value the entire stock of the asset (the whole protected area), which often generates multi-trillion-dollar estimates⁹⁴⁻⁹⁶. Rather, we calculated the ‘value of protection’. For example, protected areas might reduce the loss of tropical forest, implying a reduction in the loss of forest-associated economic values. The value of protection is therefore the difference between the change in value under business as usual (the no-PA-expansion scenario), and the change in value under enhanced protection implied by the 30% target. Thus, for mangroves, we used the approach of Brander et al⁴, estimating where mangroves might be lost and how additional protection would alter that loss. For tropical forest values, we similarly calculated the avoided loss values using the spatially explicit value grid of Carrasco et al³⁴. And for carbon, we again compared the change in forest carbon stocks under PA expansion and under no-PA-expansion, using the Global Forest Products model (see methods appendix)⁶⁹. The marginal value of protection is obviously much smaller than the value of the entire protected area (i.e. the value of the stock^{97,98}).

Our results suggested that even for that highly incomplete subset, the marginal value of the 30% proposal could be considerable, ranging from \$170 billion to \$530 billion annually (Table 3). This is by definition an underestimate because it does not consider any benefits from the protection of other ecosystems such as temperate woodlands, tundra or unmanaged grasslands, nor emerging services such as reducing the risk of highly costly pandemics^{35,99}. The benefits may also be larger if the demand for nature’s services increases, as could happen if there is large localised population growth in an area where drinking water, or the health benefits of nature, partly depend on the presence of natural ecosystems.

Finally, we considered some of the non-monetary benefits of enhanced nature protection that would be difficult or inappropriate to cost, even with environmental accounting tools. Most obviously, the proposed expansion of PAs is designed to have a major positive impact on global biodiversity, and a PA expansion of the magnitude proposed would be expected to lower extinction risks and preserve natural habitats (we are currently in the process of calculating the biodiversity benefit measures for the scenarios). Our scenarios would also increase the area recognised as representing the stewardship of Indigenous Peoples and local communities by between 63% and 98%, depending on scenario. This is likely to be a positive outcome for any communities whose cultural or livelihood preferences align closely with conservation objectives, so long as governance arrangements maintain an appropriate rights framework, for example by creating or identifying OECMs rather than traditional PAs where IPLC express that desire, recognising IPLC as central knowledge-, rights- and stakeholders, and indeed by involving them strongly in the decision-making processes about whether an area should be formally protected at all (based on the principle of Free Prior Informed Consent)^{19,38}. As we emphasised in our discussion on opportunity costs, having legal recognition of the need to preserve natural land can either be a benefit to communities who traditionally own, manage, use or occupy such land, or it can be a cost, depending on the governance and rights approach taken and the level of financial and non-financial support that IPLC stewardship of the land attracts from governments.

Costs and Investment Needs for the 30% Target

METHODS: BUDGET NEEDS (COSTS)

Our budget-need calculations for implementing the 30% target are all based on analysing bottom-up estimates from country parties of the minimum adequate budget needs for their protected area systems. These bottom-up estimates were largely drawn from the Financial Scorecards submitted to the United Nations Development Program (UNDP) on PA system needs¹⁰⁰, complemented by extensive research into other statistics and reports available (Methods Appendix). We then used a statistical model of those budget needs to predict the costs of an expanded system. Our cost models account for differences in system size, pressure, level of land competition and cost, and remoteness, since these are the factors that were shown to influence existing budget needs (Methods Appendix).

We specify *a priori* that for a PA network, the cost/investment estimate reflects the cost of a minimum adequate level of management. Many PAs do not currently receive enough funding to meet that minimum level, causing them to fail in their basic mission of protecting the country's basic natural infrastructure and biodiversity. For example, Gill et al. found that shortfalls in staff and budgets were the most important factors driving the widespread failure of global MPAs to achieve basic conservation objectives¹⁰¹ and Coad et al. found that only one in four PAs have adequate resources to properly function (which, if unaccounted for, might lead to an overestimate of the global level of species protection of up to 700%¹⁰²). On a more economic note, there is also extensive research showing that nature tourism can suffer permanent losses in revenue flows if the protected area is not maintained in a high-quality state. For example, the poaching of elephants in under-resourced African parks has a detrimental effect on tourism worth \$25 million per year¹⁰³, and two thirds of visitors to India's most frequented protected areas state that they would not come if they could not see tigers any more (e.g. due to poaching¹⁰⁴). Our cost estimates therefore include both the minimum adequate budget need for existing national PA networks, and the minimum adequate budget need for an expanded PA network.

In addition to the annual management costs, there may be additional budget lines to purchase land and place protective infrastructure on it (e.g. buildings and transport for a ranger force). There is a dearth of literature that would allow us to estimate the infrastructure spending needs for establishing a new

protected area system, especially for a large-scale exercise with multiple options about where PAs might be located. Our costing models include minimum adequate capital spending needs (CAPEX, including infrastructure), again derived from countries' own estimates. However, those bottom-up estimates gave insufficient detail to allow us to account for possible differences between CAPEX needs in existing and newly-established PAs. This will cause an underestimate in annualised budget figures, depending on the magnitude and timescale of any infrastructure construction needed in new PAs. Land purchase costs are similarly difficult to estimate because there is no global database of land prices and indeed, many conservation areas could be established on land where purchase is legally and morally inappropriate (e.g. government-owned or IPLC lands). A common proxy for agricultural land price is the net farm income divided by the capitalization rate of agricultural land (where income has to be modelled based on the expected production and costs of not-yet-converted land¹⁰⁵). We calculated this proxy for all the new land in each scenario and amortised over 20 years. We note that this will give an overestimate, since it ignores yield gaps^{106,107}, cost savings on government or Indigenous lands, and also cost savings from large-scale industrial farming. Also, the capitalization rates are taken from studies of European farms, and are likely to also overestimate costs in non-EU countries. Thus, CAPEX is underestimated and purchase cost is overestimated, partially cancelling out their respective errors in the calculation of establishment cost.

The final cost is compensation. Even if the national agricultural sector can simply readjust its pattern of expansion to take account of protected areas, PA creation can have immediate economic costs for local people who would not necessarily be able to move to the 'next best available place'. For equitability and to avoid political resistance, protected area establishment should therefore ideally compensate local peoples for livelihood and other losses, and indeed to take all appropriate steps to ease the economic transition, otherwise local economic losses can greatly reduce the success of protected area initiatives¹⁰⁸⁻¹¹². We defined two types of local opportunity costs: a lost-production cost for actors who earn income or have subsistence livelihoods based on agricultural cultivation; and a natural-resource-user cost, for actors such as IPLC who may often use the natural resources of a protected area to fulfil their material and non-material needs¹¹³. Lost-production costs will need compensation in proportion to the value of the land. For simplicity and full compensation, we used the full land purchase cost. To convert this to an annual value, we divided up the value over 10 years and added it to other costs already calculated. Some of the scenarios may also have had important effects on human-wildlife conflict (for example, the Global Deal for Nature explicitly considers the needs of wide-ranging carnivores that can have impacts ranging from economic damage to loss of life, and the Biodiversity/Wilderness Consensus similarly envisages a hybrid of traditional and landscape-level or less formalised approaches to conservation areas¹¹⁴). However, there was no robust way of estimating how the six global scenarios might have led to a change in compensation costs for human-wildlife conflict incidents. We limit ourselves to observing that setting aside funds for compensation, with the amount depending on how far the implementation of the 30% target risks increasing conflict, would be an important element when putting the policy into action.

Our non-market type of opportunity cost reflects the potential risk for IPLC who depend on natural resources for their subsistence, including IPLC who already and currently inhabit possible future conservation areas. Non-market opportunity costs would arise (and have sometimes arisen in the past) if conservation regulations result in IPLC displacements and restrictions over access and use rights to land and resources, harming not just their livelihoods but also their cultures, histories and identities^{38,115}. However, if governance arrangements permit and respect customary uses of biodiversity by such groups – as is recommended in the Global Biodiversity Framework – then such non-market opportunity costs are largely minimized. Indeed, under rights-based governance approaches, the presence of a conservation area (probably in the form of an OECM or an Indigenous and Community Conserved Area (ICCA)) can be a benefit rather than a cost, for example by strengthening IPLC land-use rights. Being non-monetary, potential resource-user opportunity costs are not included in the budget estimates, but the Global Biodiversity Framework emphasises that they need to be carefully borne in mind.

RESULTS: BUDGET NEEDS (COSTS) AND COMPARISON WITH CURRENT SPENDING

The final estimated budget or investment need for the 30% target was between \$103 billion and \$177.5 billion per year, depending on the scenario used (Table 3). This cost is divided up into the minimum budget needed to adequately manage the existing PA system (\$67.6 billion per year) and the cost of adding new areas to reach the 30% target (between \$35.5 billion and \$110 billion per year)⁵. Scenarios that strongly prioritize production tended to have lower costs (as well as lower revenues) (Figure 5). The majority of the annual cost relates to management expenses (including both operating costs and capital expenditure), with relatively small values for annualised amortised land purchase costs and possible compensation costs, e.g. for local farmers who are unable to expand their fields. Between 69% and 91% of that cost would be for implementation in low- and middle-income countries (“developing countries”). This may be a slight overestimate because the level of PA underfunding in high-income (“developed”) countries is not well assessed, and so the developed-country budgets needed may be higher than modelled. Even so, the proportion of costs needed for conservation in developing countries would be considerable.

For comparison, we estimate that current global spending on PAs is ~\$24.3 billion per year (calculated as \$17.2 billion collated from primary sources for 124 countries, \$4.5 billion invested by local communities¹¹⁶, and \$2.6 billion imputed for countries with no data, see Methods Appendix). The previous global estimate of spending on PAs (in 2001) was \$6 billion per year¹¹⁷. The earlier figure is not strictly comparable with the later one, owing to differences of method, but the difference does suggest a broad, consistent political willingness to increase spending on the protection of nature, as also evidenced by our analysis of biodiversity aid increases (see Methods Appendix).

Some caveats on these budget need estimates are required. First, there is almost no assessment available for the minimum adequate budgets required by PAs in high-income countries, even though reports on the EU and North America both suggest an uncertain degree of underfunding there^{118,119}. Our budget-need estimates therefore had to assume that the current budgets per hectare could be used to estimate the costs of an expanded and adequately funded system. High-income countries will therefore need to carry out their own assessments, and the results may increase the global budget need. Conversely, future budget needs may be lower than stated if management efficiency in protected area networks increases, and this may be especially true of developing countries (where percentage efficiency gains can be largest).

Thirdly, there is little data on which to robustly estimate, at a global level, any difference between the costs of standard or traditional PAs (which was the main focus of bottom-up budget needs reported by countries) and the costs of OECMs or PAs under local co-management. Some case studies suggest that ICCAs (territories conserved by indigenous peoples and local communities) may have lower running costs (not least because IPLC can themselves make substantial investments in protection¹¹⁵), and similar budget advantages may occur with other types of community management^{115,120-122}. But there are also clear moral and practical risks if it is assumed that all such territories and communities worldwide would need less financial support than is given for standard PA governance approaches. For example, some ICCAs may come under considerable pressure (despite their technical legal status) and may require strong guardianship assistance at high cost.

⁵ The ranges for both total costs and additional costs do not include an unusually high estimate for the Global Deal for Nature (Table 3), see comments above on how that scenario has a strong landscape element that probably cannot be robustly modelled from existing data on more traditional PA costs.

Finally, one particular scenario (the Global Deal for Nature or GDN) appeared to have a much higher cost than any other scenario, exaggerating the general sense of how much the 30% target might require as a budget. However, this cost is likely to be an artefact of an unusually high landscape-level focus in that scenario. The GDN, when mapped as a sense of spatial priorities, includes large tracts of valuable agricultural land, triggering high estimates of both land costs and compensation needs. However, the spirit of that scenario is not that existing agricultural land should be taken inside a traditional PA, but rather because some parts of the landscape are necessarily included to satisfy sub-priorities such as the ranging needs of large carnivores. The bottom-up estimates of PA system costs (which underpinned our costing model) are likely to be based on more traditional PA management approaches, causing a cost overestimate for landscape areas that do not need traditional forms of protection. Other scenarios (such as the Biodiversity/Wilderness Consensus) may also have lower-than-estimated costs for similar reasons. It is more difficult to judge whether the estimates of benefits may be affected by landscape-level approaches, but our overall conclusions would not be changed by the exclusion of landscape-approach scenarios.

Synthesis: Putting Together the Benefits and Costs of Investing in the 30% Target

To date, policymakers have often seen conserving nature as a non-financial benefit, sometimes described as a “nice to have” rather than an engine of the economy. Indeed, protected areas were often felt to be a brake on economic growth. This report has shown how protected areas not only contribute strongly to economies, but already enjoy some of the fastest-growing revenues of any sector.

However, protected area expansion could still generate losses to other industries (for example, if a new PA obliges agriculture to expand onto its second-choice area rather than the first-choice one). This balance of gains and losses required an economic analysis across the multiple sectors affected, asking whether the economic bottom line, when all the sectors are added together, would benefit more from expanding PAs and their revenues, or from not expanding them⁵⁵⁻⁶⁰.

We found that when we measured the financial value of outputs^{19,120} across combined sectors, expanding the PA sector and its revenues resulted in a modest but positive outcome for the global economy. If incorporating multiplier effects, the increase in gross expenditure (output) is in the projected range of less than one hundred billion dollars per year to over one trillion dollars per year, depending on the implementation scenario and multiplier value used. This projected increase was partly driven by growth in global visitor numbers, but was robust to the threat of future reductions in tourism. Sub-analyses where we incorporated potential increases in agricultural producer expenditure (without considering any benefits that other actors may receive when that expenditure occurs) suggested a smaller financial benefit. In purely financial terms, our analysis therefore broadly indicates that the 30% target is expected to be either neutral or modestly positive, depending on which economic indices are used.

If expansion and non-expansion have similar financial outcomes (with perhaps a modest gain under expansion), then the ultimate decision would likely depend on the broader economic and social consequences of the proposed 30% target. Here, our analyses would suggest that PA expansion has a strong advantage. Firstly, expansion strongly outperformed non-expansion on risk mitigation and avoided cost values, including the potentially very large economic risks of climate change, biodiversity loss and emerging diseases^{1,2,123,124}. Expansion would also help governments meet their commitments to many global priorities, including the Paris Agreement on greenhouse gas emissions and climate change, the Sustainable Development Goals for human wellbeing, sustainable development and biodiversity, and the CBD mission to halt species extinctions^{37,125,126}. If some expansion takes the form of OECMs, it also offers an opportunity to strengthen IPLC customary land use rights where these align with biodiversity protection goals, provided that full prior informed consent and a full rights framework are guaranteed.

The expansion/non-expansion decision will also be affected by the relative prospects for future economic growth. In this respect, an important finding is that the PA/nature sector has a considerably higher revenue growth rate than agriculture, forestry, fisheries, or overall global GDP (e.g. wildlife tourism is estimated to be growing at 10% per year in Africa⁴⁵). There is also considerable new tourism potential as developing-country source markets become wealthier (leading to growth in South-South and regional tourism) and as human populations become increasingly concentrated in cities, enhancing demand for large natural areas. The economic multipliers estimated for tourism (including PA tourism) are also higher than those estimated for the forestry, agriculture and fisheries sectors^{20-22,24,127}, increasing the economic value of any new revenues generated by PA expansion.

(Given current travel restrictions from COVID-19, we comment that the tourism sector's current 4% growth average includes a prolonged period of travel stagnation from 2001-2003 due to terrorism (including the 9/11 attacks), the global SARS epidemic, and the 2007-9 global economic crisis¹²⁸. This experience suggests that the tourism industry still has the potential to grow post 2030. However, precise behavioural and trade changes are hard to predict.)

The fastest revenue growth would occur if we protect places near larger or wealthier human populations, since those are the areas where tourism is more frequent and ecosystem service values are highest (due to greater local demand). The implication is that, under the new nature economy, it will sometimes make more economic sense to locate PAs near to people, where they can provide the most benefits, rather than the traditional practice of placing them in often-remote areas of low importance to human production^{86,87}. Creating national landscapes in which people and nature are more closely associated could have multiple benefits, including the health and productivity benefits that have been documented when human beings are able to access natural and forested areas^{36,51,52,129,130}. However, it would also require a strong impetus for sustainability (in the Western idiom) or living in harmony with nature (eco-civilisation in the current Chinese policy framework³¹). Initially, sustainability may have to be imposed, temporarily raising the costs in exchange for the benefits of future revenue growth.

We caution that our analysis is likely to have underestimated the future benefits, both non-monetary and financial. Markets are already beginning to innovate new revenue streams for nature, and these have very rapid growth (for example, REDD+ offsets grew by 187% in the latest reporting period for carbon markets¹³²). Implementing the proposed 30% target would itself be expected to boost growth in the PA/nature sector even further, because almost one third of the planet's asset base by area would be physically located in that sector, and because the creation of new PAs implies a further expansion of nature tourism revenues.

We may also have overestimated the costs, especially in lower-income countries, because we were unable to account for the future impact of improvements in management efficiency, the fact that much of the 30% target may be achieved by creating OECMs (which have potentially lower costs that remain undefined), and potential cost savings from increased IPLC and community management involvement. If we also achieve a more sustainable economy¹ outside PAs, this would also reduce costs for the 30%, by mitigating pressures. Conversely, greater pressures in areas that contain PAs may increase the costs of conservation¹³³.

In public investment analysis, an estimation of costs and benefits for only the sector invested in is often required^{134,135}. If we consider the PA sector on its own, our results consistently suggest the 30% proposal would yield benefits several times larger than its investment costs, both in revenues and in broad-sense economic terms (i.e. public goods) (Figure 5). Various levels of investment ambition are possible for PAs but in general, higher investment generates higher financial and economic benefits (Figure 5). The major constraint on spending for nature is not therefore, the apparent non-productivity of nature, but the degree to which funders can afford to invest more in order to achieve larger rewards.

Although nature tourism can financially incentivise and support protected areas, an obvious objection is that excessive tourism pressures inside natural areas can cause harm¹³⁶. Sustainable tourism studies emphasise a clear trade-off between allowing tourism to fund and help protect PAs, and allowing market forces to dominate so extensively that damage occurs^{137,138}. However, an additional benefit of greatly increasing the protected area estate is that if new destinations are developed, it could draw some of the worst crowding away to other areas, potentially reducing the pressure on the most crowded individual sites.

DISTRIBUTION OF COSTS AND BENEFITS, DEVELOPMENT ASSISTANCE, AND CROSS-SUBSIDY

Although the overall picture shows a net financial and economic benefit from expanding PAs, the distribution of the benefits and costs will not be homogeneous, raising several important issues. First, land-use or ocean-use planning decisions typically cause benefits to some economic sectors but costs to others. Second, a net positive outcome globally does not necessarily indicate a positive outcome for each individual country, at least at current levels of development. Lower-income countries may often lack the necessary infrastructure and market access to realise the revenue potential from PA expansion, removing the incentive to participate in the costs of the global initiative. Such countries are likely to be left out of the fast-growing revenue opportunities, exacerbating global economic inequality. To avoid this outcome, such countries would require development assistance, to help them capitalise on the economic value of their natural assets (in which they often have a competitive asset-volume advantage).

Heterogeneous negative effects can also occur at local and individual scales. For example, if a protected area covered all available unconverted land for a smallholder community, then it could cause considerable local opportunity costs, even if the outcome at national level was an economic gain. Local opportunity costs can generate considerable social and economic problems, and past experiences show that dissatisfied local actors can engage in long-term opposition and resistance to a protected area, when they could be supporters and co-managers under rights-based approaches and inclusive forms of governance^{116,139,140}. To minimise these negative effects, cross-subsidy and broader livelihood support, compensation and outreach may be needed. In many countries, PA tourist revenues are already redistributed to provide support, infrastructure and development for local communities, smoothing the transition to a more nature-based local economy¹⁴¹⁻¹⁴⁵. But in general, the transition towards a more nature-based economy would need to be carefully managed and communicated, acknowledging distributional inequalities and mitigating them to preserve the overall global benefit.

TABLE 3.

Benefits and costs of scenarios to expand protected areas (2050). See Table 2 for glossary of scenario-name abbreviations.

All figures are in billions of US dollars at 2015 values

SCENARIO OPTIONS									
	REF (No PA expansion)	THC	HPR	BIWI	BPC	SSE	GDN		
MULTI-SECTOR ANALYSIS									
FINANCIAL VALUES (gross outputs = revenues) USD bns									
PA/Nature sector output value: nature tourism (terrestrial)	190.85	190.85	190.85	190.85	287.17	244.00	287.17		
PA/Nature sector output value: nature tourism (marine: coral reef associated only)	130.22	140.14	140.14	146.82	150.40	196.89	196.89		
PA/Nature sector output value: private flows	50.00	50.00	50.00	60.00	60.00	60.00	60.00		
Agricultural output value 2050 (mean of models weighted by output magnitude)	4,399.32	4,434.56	4,547.47	4,497.56	4,474.25	4,536.95	4,580.69		
Agricultural output value (max of models)	4,757.92	4,756.79	4,843.22	4,756.79	4,760.24	4,753.61	4,830.49		
Agricultural output value (min of models)	4,006.30	4,046.53	4,046.49	4,043.79	4,049.80	4,113.23	4,157.75		
Fisheries output (BOATS model)	33.77	23.37	23.37	27.05	35.78	24.81	24.81		
Forestry output value	427.92	449.73	467.80	506.81	459.63	511.32	528.83		
Aid: Biodiversity	7.39	14.78	14.78	14.78	14.78	14.78	14.78		
Aid: agriculture	12.00	12.00	12.00	12.00	12.00	12.00	12.00		
Aid: fisheries	0.08	0.08	0.08	0.08	0.08	0.08	0.08		
Total PA/Nature Sector	378.46	395.77	395.77	412.45	512.35	515.67	558.84		
Total Agricultural Sector	4,411.32	4,446.56	4,559.47	4,509.56	4,486.25	4,548.95	4,592.69		
Total Fisheries Sector (wild-capture only)	33.85	23.45	23.45	27.13	35.86	24.89	24.89		
Total Forestry Sector	427.92	449.73	467.80	506.81	459.63	511.32	528.83		
Total output/revenue across all sectors analysed	5,251.54	5,315.51	5,446.50	5,455.95	5,494.09	5,600.84	5,705.24		
Total output/revenue difference (increase) compared to no-PA-expansion baseline	NA	63.97	194.96	204.41	242.55	349.30	453.70		
Total output/revenue % change compared to no-PA-expansion baseline	NA	1.2%	3.7%	3.9%	4.6%	6.7%	8.6%		

SINGLE-SECTOR ANALYSIS: PA sector only										
COSTS (USD billions)										
Annualised cost to add new protected areas up to 30% target	0.00	35.50	44.42	109.89	70.54	103.30	223.83			
Annual costs to give minimum adequate funding to existing protected areas	67.64	67.64	67.64	67.64	67.64	67.64	67.64			
Total costs/yr	67.64	103.13	112.05	177.53	138.17	170.94	291.47			
REVENUES (USD billions)										
Revenues	378.46	395.77	395.77	412.45	512.35	515.67	558.84			
Revenues with 3.2 multiplier applied to tourism component	1,084.80	1,123.96	1,123.96	1,155.32	1,475.00	1,485.62	1,623.76			
ECONOMIC VALUES (avoided cost values compared to baseline); USD bn; 2050 snapshot										
Tropical forest ecosystem service values \$bn/yr	NA	96.80	101.93	103.98	104.12	136.66	142.20			
Mangrove ecosystem service values \$bn/yr	NA	53.91	53.91	5.27	79.92	79.45	79.45			
Forest carbon values \$bn/yr	NA	19.71	89.84	276.67	92.43	293.18	312.71			
Total economic value \$bn/yr	NA	170.42	245.68	385.92	276.47	509.29	534.37			
REVENUES PLUS ECONOMIC VALUE; \$bn/yr, 2050 (without multiplier effects)	378.46	566.20	641.45	798.36	788.82	1,024.96	1,093.21			
COMPARISON OF SINGLE-SECTOR PA COSTS AND REVENUES (but see Synthesis for caveats); 2050										
Revenues minus costs	310.82	292.64	283.72	234.92	374.17	344.73	267.37			
Revenues plus economic value minus costs	310.82	463.06	529.40	620.84	650.65	854.02	801.73			
Multiplier-adjusted revenues minus costs (without economic values)	1,017.16	1,020.83	1,011.91	977.79	1,336.82	1,314.68	1,332.29			
SECTOR-SPECIFIC CHANGES IN GROSS OUTPUT CAUSED BY IMPLEMENTING THE 30% TARGET										
difference in gross output: PA/Nature Sector \$bn	NA	17.32	17.32	33.99	133.89	137.21	180.38			
difference in gross output: Agriculture \$bn	NA	35.24	148.15	98.24	74.93	137.64	181.37			
difference in gross output: Forestry \$bn	NA	21.81	39.89	78.89	31.71	83.41	100.91			
difference in gross output: Fisheries \$bn	NA	-10.40	-10.40	-6.72	2.02	-8.96	-8.96			

AUTHOR CONTRIBUTIONS

Anthony Waldron (overall project design, coordination, analysis and writing), Vanessa M. Adams (scenarios and writing), James R. Allan (scenarios), Andy Arnell (current protected areas and GIS), Gregory P. Asner (scenarios), Scott Atkinson (scenarios), Alessandro Baccini (scenarios), Jonathan E.M. Baillie (scenarios), Andrew Balmford (tourism analysis and data, writing, design), Beau J. Austin (IPLC), Luke Brander (marine economic valuation), Eduardo Brondizio (IPLC), Aaron Bruner (writing/economics), Neil D Burgess (writing and policy), Karl Burkart (scenarios), Stuart Butchart (writing, policy and scenarios), Rio E. Button (GIS), L. Roman Carrasco (agricultural economic data and economic valuation), William Cheung (fisheries analysis), Villy Christensen (fisheries analysis), Andy Clements (scenarios), Marta Coll (fisheries analysis), Moreno Di Marco (scenarios and writing), Marine Deguignet (current protected areas), Eric Dinerstein (scenarios), Erle C Ellis (scenarios and IPLC), Florian Eppink (marine economic valuation), Jamison Ervin (project design and policy), Anita Escobedo (GIS), John E. Fa (scenarios), Álvaro Fernández-Llamazares (IPLC and writing), Sanjiv Fernando (scenarios), Shinichiro Fujimori (agricultural analysis), Beth Fulton (marine analysis, design and writing), Stephen T. Garnett (IPLC and writing), James Gerber (agricultural analysis and data), David Gill (marine analysis and data), Trisha Gopalakrishna (forestry analysis and GIS), Nathan Hahn (scenarios), Benjamin S. Halpern (marine scenarios and writing), Tomoko Hasegawa (agricultural analysis and writing), Petr Havlík (agricultural analysis and writing), Vuokko Heikinheimo (social media and tourism analysis), Ryan Heneghan (fisheries analysis), Ella G. Henry (marine protected area analysis), Florian Humpenöder ((agricultural analysis and writing), Harry Jonas (OECMs, policy and writing), Kendall R. Jones (scenarios), Lucas Joppa (scenarios), Anup Joshi (scenarios), Martin Jung (scenarios and programming), Naomi Kingston (protected area policy, writing), Carissa Klein (scenarios), Tamas Krisztin (agricultural analysis), Vicky Lam (fisheries analysis), David Leclère (agricultural analysis, writing, economics, coordination), Peter Lindsey (data and writing), Harvey Locke (scenarios, project design), TE Lovejoy (scenarios), Philip Madgwick (data collection and analysis), Yadvinder Malhi (forestry analysis), Pernilla Malmer (IPLC and writing), Martine Maron (scenarios and writing), Juan Mayorga (scenarios), Hans van Meijl (agricultural analysis), Daniel C. Miller (financial flows data, policy, writing), Zsolt Molnar (agricultural analysis), Nathaniel D Mueller (agricultural analysis and data), Nibedita Mukherjee (data collection), Robin Naidoo (writing and policy), Katia Nakamura (GIS and data collection), Pablo Jose Negret (GIS), Prakash Nepal (forestry analysis), Reed F. Noss (scenarios), Bethan C. O’Leary (design, policy and writing), David Olson (scenarios), Juliano Palacios-Abrantes (fisheries analysis), Midori Paxton (financial data and policy), Alexander Popp (agricultural analysis), Hugh Possingham (scenarios and writing), Jeffrey P. Prestemon (forestry analysis), April E. Reside (scenarios), Catherine Robinson (scenarios), John Robinson (policy and writing), Enric Sala (scenarios, policy and writing), Kim Scherrer (fisheries analysis and writing), Mark Spalding (marine economic data), Anna Spenceley (tourism and writing), Jeroen Steenbeek (fisheries analysis and writing), Elke Stehfest (agricultural analysis), Bernardo Strassburg (economic design and writing), Rashid Sumaila (fisheries analysis and writing), Kirsty Swinnerton (data collection), Jocelyne Sze (GIS), Derek Tittensor (fisheries analysis and coordination), Tuuli Toivonen (social media and tourism analysis, writing), Alejandra Toledo (data collection), Willem-Jan Van Zeist (agricultural analysis), James Vause (economic design and writing), Oscar Venter (scenarios and data), Thais Vilela (GIS), Piero Visconti (design and scenarios), Carly Vynne (scenarios), Reg Watson (fisheries data and analysis), James Watson (scenarios and writing), Eric Wikramanayake (scenarios), Brooke Williams (scenarios), Brendan Wintle (scenarios), Stephen Woodley (scenarios), Wenchao Wu (agricultural analysis), Kerstin Zander (IPLC), Yuchen Zhang (agricultural data), Y.P. Zhang (scenarios)

FUNDING ACKNOWLEDGMENTS

A. Waldron, K. Nakamura, J. Sze, T. Vilela, A. Escobedo, P. Negret Torres, R. Button, K. Swinnerton, A. Toledo, P. Madgwick, N. Mukherjee were supported by National Geographic and the Resources Legacy Fund. V. Christensen was supported by NSERC Discovery Grant RGPIN-2019-04901. M. Coll and J. Steenbeek were supported by EU Horizon 2020 research and innovation programme under grant agreement No 817578 (TRIATLAS). D. Leclere was supported by TradeHub UKRI CGRF project. R. Heneghan was supported by Spanish Ministry of Science, Innovation and Universities, Acciones de Programacion Conjunta Internacional (PCIN-2017-115). M. di Marco was supported by MIUR Rita Levi Montalcini programme. A. Fernandez-Llamazares was supported by Academy of Finland (grant nr. 311176). S. Fujimori and T. Hawegawa were supported by The Environment Research and Technology Development Fund (2-2002) of the Environmental Restoration and Conservation Agency of Japan and the Sumitomo Foundation. V. Heikinheimo was supported by Kone Foundation, Social Media for Conservation project. K. Scherrer was supported by the European Research Council (ERC) under the European Union's Horizon 2020 research and innovation programme under grant agreement No 682602. U. Rashid Sumaila acknowledges the OceanCanada Partnership, which funded by the Social Sciences and Humanities Research Council of Canada (SSHRC). T. Toivonen was supported by Osk. Huttunen Foundation & Clare Hall college, Cambridge. W. Wu was supported by The Environment Research and Technology Development Fund (2-2002) of the Environmental Restoration and Conservation Agency of Japan. Z. Yuchen was supported by a Ministry of Education of Singapore Research Scholarship Block (RSB) Research Fellowship.

METHODS APPENDIX:

Protecting 30% of the Planet for Nature: Costs, Benefits and Economic Implications

Outline methods for the Working Paper can be found in the Full Report section of the main report. The Methods Appendix gives further details where they may be useful.

REVENUES/SECTOR OUTPUTS

PA/Nature Sector

Calculating the value of nature tourism involves three basic steps: (1) estimate the number of visitors (or visits) to each country's protected area system; (2) estimate the average revenue that is generated by each visitor or visit; (3) create a model projection of post-2030 visits and associated revenue expectations.

To estimate visitor numbers, we first collated all statistics on the number of visitors to national protected area systems, by searching government statistics, reports by governments and development agencies, and primary academic research, in multiple languages (English, French, Portuguese, Spanish, German and Bahasa Melayu/Bahasa Indonesia, plus Russian with the assistance of a translator). We were able to find data for 118 countries. We carried out similar research to collate data on visitor spending, building on the database of Balmford et al⁴⁴. In some cases, the data reported covered the main tourist sites rather than all PAs in a country, but we assume that the error introduced by such gaps would be minor with respect to the overall pattern of visits. Some countries have no published data (that we could find) on their visitor statistics. We therefore treated the data we collated as a representative sample and built a set of predictive models based on factors known to influence visits.

We hypothesized the following factors might influence the annual number of protected area visitors to a national PA system: the total number of foreign visitors ("international arrivals") entering the country in that year, showing the size of the international tourist pool from which PA-specific tourism could draw; the mean accessibility of the PA system, reflecting ease of travel from the major cities where both international and domestic tourists are most likely to originate their journeys from; the population density in the areas surrounding the PAs (measured in a 25km buffer around all PAs in each country), reflecting the size of the pool of potential local visitors; the GDP per capita (in the year of the data source), reflecting the likelihood that wealthier individuals have greater capacity to travel and spend money in protected areas; the year to which the data referred, since visitor numbers have tended to go up over time; governance, since instability or poor administration may deter visitors (especially international ones); the percentage of the total national area that is a formally registered protected area, because the more of the country that is a protected area, the more likely it is that any tourist will visit one as part of their tour; the IUCN protection category, since PAs in category I do not allow visitors and PAs in categories V and VI are more designed for community use of natural resources than for the recreational appreciation of nature; and a dummy variable that described nature was the dominant driver of visits to the country (defined as the country being listed as a major ecotourism or wildlife destination and also having few other tourist attractions), since this again increase the probability that any given international arrival will visit a PA.

We then created a set of candidate statistical models, each containing a subset of the hypothesized predictors, to model known PA visitor numbers. We used Generalized Additive models (GAMs)¹⁴⁶ because PA visitor numbers had a non-linear relationship to international arrivals (an increase that then decelerated towards an upper asymptote). PA visitor numbers, international arrivals and GDP per capita had strongly skewed frequency distributions and so we

In-transformed them. We also set the gamma parameter for the GAMs to 1.5, in order to prevent overfitting (which can increase the predictive power of the model on the data in hand but reduce the generalizability of the model to predict unknown cases). We theorized that population density and the population's wealth (GDP per capita) might interact in the way they influence the number of visitors, so we tried fitting a statistical interaction to those terms. We used the Information Theoretic criterion adjusted for small sample sizes ("AICc", or Akaike's Information Criterion)¹⁴⁷ to assess relative goodness of fit of a number of models that contained subsets of the theorized predictors, then examined the R-squared and P values post hoc to assess absolute predictive power.

The best-fit model explained 89.3% of the variation in the data (n=58, adjusted R-squared = 89.3, variance explained = 91.1%) and all variables had P-values of <0.05 in the post-hoc examination. As expected, PA visitor numbers increased non-linearly as international arrivals increased. Countries with greater GDP per capita and higher population density in the PA buffers had more PA visitors, and these effects interacted, with higher levels of one variable increasing the influence of the other variable; (in simple terms, places with a large pool of potential local tourists tended to have more PA visitors if people in that pool were wealthier). Countries with more accessible PA systems had higher visitor numbers. The year the tourist data were taken from had the expected impact, with more recent years having higher tourist numbers. Additionally, countries where the entire tourist economy was known to be nature-based generally had higher PA visitors than expected, all else being equal. For the European Union, a detailed, EU-specific study had already generated its own model and used it to generate an estimate of total visitors numbers for most European Union countries (Schagner 2017)^{148,149}. We therefore omitted European countries from our own modelling and used Schagner's estimates of visitor numbers instead, noting that these are likely to be underestimates because they focus on National Parks.

Given the high predictive power of our generalized model (and of Schagner's equivalent model for EU visitors), we used the model to estimate the tourist numbers in non-EU countries where there was no information on visitor numbers to their national PA system. We also standardized all visitor values to a common year of 2017. If a known data point predated 2017, we predicted it forward by changing the year of the GDP per capita data and international arrivals data to 2017, and then using those parameter changes to predict 2017 values.

We then needed to forecast nature tourism potential up to 2050, starting from the basis of this 2017 visitor data. To do this, we analysed all global social media postings that were traceable to protected area visits in each year from 2005 to 2018 and aggregated the results by country and year. Flickr is the only social media platform that yields such a long time series of publicly analysable data¹⁵⁰ and has also been shown to be a good predictor of officially collected visitor statistics¹⁵¹, so we focused on that platform. Thus, we used data social media platforms as a proxy for visitor numbers in protected areas. We accessed data from the Flickr API (<https://www.flickr.com/services/api/>, accessed in Spring 2019) using the Python programming language and considered data for years 2005 - 2018. We used polygons from the World Database on Protected Areas (WDPA) data set from October 2019 (<https://www.protectedplanet.net/>) in the social media calculations. We calculated the count of unique users per protected area per day and divided the number by the count of unique users per country per day for each protected area, in order to correct for changes in the number of social media users each year. We then used these normalized visitor counts in a time series model to generate a forecasting model for protected area visits post-2020. We tested both a cubic spline analysis and a damped exponential smoother, using known rates of recent growth in the nature tourism sector to assess the performance of the different approaches on the data.

Tourism in many countries has been quite volatile between 2005 and 2018, so a conservative approach is warranted, but it is also important not to allow extreme short-term shocks (e.g. spikes in terrorism) to exert undue influence on the forecast across multiple future years. Observed long-term rates of international tourism growth are about 4% and nature tourism may be increasing faster than the overall industry average^{45,128,152-154}. Additionally, the coefficient for the year term in our statistical model was 6%. In exploratory testing, the cubic spline predicted 4.4-5.2% growth (depending on period assessed) and the exponential damped smoother predicted 2.2%-2.6%. Since the cubic spline reflected the current rate of (pre-COVID-19) growth and the damped exponential smoother predicted only half the empirically observed growth, we report the results using the cubic spline in the main analysis. We note that the splines' projected compound rate of annual growth in visitors/postings also decreased over time, suggesting that the splines were not predicting infeasible levels of growth into the future.

To convert visitor numbers into an economic value based on their level of spending around the protected area, we used the approach of Balmford et al.⁴⁴, multiplying visitor numbers by mean visitor spending. Where an official source reported visitor spending per year in the protected area system, we used that information. But for countries where this information was unavailable, we followed the methods of Balmford et al. (2015)⁴⁴, by taking the median value of all spending-per-visitor statistics we found reported in the literature for that region, and applying it to impute the datapoints for the missing countries in that region. Schagner's EU study applies a single value for visitor spending across the entire EU (based on a meta-analysis), but our data contained several values for individual European countries, and so for consistency we used country-specific European values calculated from our data.

We also compared our estimated visitor statistics by country for 2017 to Balmford's original table of visit statistics, which are admittedly mostly from earlier years. Even so, we noticed that in a small number of countries, Balmford's numbers could be an order of magnitude larger than ours. It seemed most likely that this occurred because some countries – particularly in developed countries – have a very large number of casual day visits to small, local protected areas, for purposes as humble as walking a dog. It was precisely in such countries that the differences between the estimates arose. The analytical importance of this difference is that the spending data is also likely to be biased towards Type I visitors, who leave a detectable economic trace, and so it would be inappropriate to assume that people who were 'walking the dog' have the same level of spending as people who have come for a weekend in a national park or bought a safari package, for example. Balmford et al. also collated data from individual PAs whereas we collated data from national PA systems. Since national PA systems tend to collate data on visitors that leave an important economic trace in the area such as overnight visitors ("Type I visitors"), whereas individual PA managers may also be recording the casual day visits ("Type II visitors"), we assumed that countries where Balmford's number of visits greatly exceeded the national statistics for visitor numbers strongly reflected Type II visitors. In our data, trip lengths for PA visits were 2-3 days on average, and some of those visits will be double counted if a visitor buys a multi-park pass or enters the same park on consecutive days. However, some Type I visitors ("Type Ib") may also be missing from national-level statistics, for example those that visit smaller provincial or state parks that lack statistical reporting systems. It would be impossible to know exactly how precisely many visitors in the discrepant countries were Type Ib and how many were Type II, despite the clear and important difference in expected spending between the categories. We therefore had to make a reasonable assumption for the small numbers of countries where the discrepancy with Balmford et al. estimates occurred. We assumed that when visits in the Balmford et al. source were more than 2.5x the national statistic for visitor numbers, a low-spending Type II visit was implied (based on the pattern in our data that trip lengths of PA visits for Type I visitors average about 2-3 days, some of which would involve multiple entries, plus the unknown number of Type Ib visitors).

To convert visits into financial values, we first multiplied Type I visitors (from national statistics) by the national-level spending-per-visitor data. For Type II visitors, a number of reports visits suggest that average expenditure would be about \$10 at purchasing power parity, for example on a light refreshment, fuel and car parking or other transport. (Some will spend more and some nothing at all, but this seems to be a reasonable value when assessing the different strands of evidence.) The purchasing power parity element converts the \$10 to a local value that would buy the equivalent set of small items in a lower-income country e.g. the same local transport, light refreshment or fuel and car parking. We then multiplied the number of each type of visitor by their expected spending in each country to derive the total contribution to national revenue that protected area visitors are expected to make. We also explored the sensitivity of our financial-value estimates to the precise ratio assumed between Type I and Type II visitors, and found that varying the ratio made little difference (owing to the very small contribution of a Type II visitor).

We then needed to apply our estimates of revenues in the existing PA system to our scenarios of an expanded system. It seems unlikely that doubling the PA system would double the visitor numbers, and so a more nuanced approach was needed. Two effects of PA system expansion would be expected to cause an increase in visitor-number growth rates. Firstly, the accessibility of a protected area affects the number of visitors^{155,156} (an observation that our statistical results for entire PA systems supports) and an important effect of declaring multiple new PAs is that for many people, access to PAs would increase. However, the precise size of this effect depends on where the PAs are located. For example, if new PAs are grouped in areas in Arctic wilderness, they still expect a relatively low number of

visitors compared to a park within easier reach of a city. We therefore calculated the difference in mean accessibility of all PAs in each scenario, and then used the accessibility coefficient from the predictive model of visitor numbers to predict the change in visitor numbers expected from the change in accessibility under the 30% target.

Secondly, creating new formally protected areas produces a brand signalling effect, whereby the fact of being formally labelled sends a motivational signal to domestic and even foreign visitors, increasing the preference experienced to visit the PA. Increases of 20% and 50% have been noted when existing green spaces have been formalised as National Parks, although a meta-analysis of the effect in the USA suggests that the mean may be as little as 6%¹⁵⁷⁻¹⁵⁹. About 14% of PAs are currently National Parks globally (in an analysis of the current WDPA global database at www.protectedplanet.net). We assumed that a similar percentage of new PAs would be designated as new National Parks and would enjoy the most conservative estimate of the visitor boost, namely 6%. However, we also assumed an interaction with accessibility, such that scenarios that created new PAs with reasonable accessibility would also benefit from the brand signalling to its full extent, whereas scenarios that often created new PAs in remote wilderness or less accessible areas (Three Conditions, Wilderness/Biodiversity Consensus and Harsh Political Reality) would not see the same signalling effect. We therefore applied 18% * 6% as the potential signalling value of new terrestrial PAs in an expanded system for Save Species from Extinction, Biodiversity/Production Compromise and Global Deal for Nature.

A more marine-focused dataset of tourism values exists in the study by Spalding et al. of how coral reefs bring visitor-based revenue into an area¹⁵⁶. These visitor values can be the main economic activity in coral reef areas, and so are important to count. There was a risk that the reef tourism values would already be included in the broader national visitor statistics, generating a risk of double counting. However, upon closer investigation, we found that in all but four cases, the countries with reef visitor values were countries that lacked data on national visitor values (many of which were small island states with primarily marine or sea's-edge tourism). We therefore treated the datasets as largely additive. We applied the growth expectations for national visitor numbers through time to the reef dataset to forecast expected reef tourism values for 2030, 2040 and 2050, then combined the reef revenues and the visitor-study revenues together to create a total global value. For the four countries where both reef and general visitor values were known and inspection of the source data could not distinguish which revenue elements belonged to which value (reef or general), we took the worst case scenario for double counting i.e. we assumed that reef revenues were already included (unless the reef revenues exceeded the total national revenue estimate, in which case we used the reef value as the national total).

Finally, we supplemented our estimates of direct tourism expenditure with an estimate of the value of PA visitor spending in the wider economy (including supply chain revenues to service the visitor and induced spending), using the World Travel And Tourism Council's average global multiplier value for travel and tourism, which is 3.2⁴³.

Biodiversity aid acts as a cash flow that enters a country due to the presence of a global public good such as protected biodiversity (contributing to GDP). If 30% of the planet were protected, it is likely that the vast majority of biodiversity would be found inside the global PA system. It is therefore not unreasonable to assume that future biodiversity aid would be strongly associated with elements of PA protection. Biodiversity aid has indeed doubled in recent years, partly in response to the call to increase PA coverage from 10% to 17% in the 2010-2020 CBD planning period. The only source of recent biodiversity aid data is the total aid flagged with the biodiversity marker in the OECD Common Reporting Standard database (CRS)¹⁶⁰, although this source is known to omit a number of donors¹⁶¹. We therefore assumed a doubling (by 2050) of the most recent aid level summarised in this source, to reach a level equivalent to about one ten thousandth of global GDP, or \$14.78 billion USD. We acknowledge that, even if a world with 30% PAs, not all biodiversity aid will necessarily be associated with the protection of nature inside that 30%, suggesting an overestimate. However, it is likely that the missing donor flows in the CRS, and indeed the estimate of a doubling by 2050, both represent potentially large underestimates. Overall, the figure given is therefore likely to be neutral in bias or more likely, a conservative underestimate. The aid component of revenues also represents <3% of the total revenue forecast, and so the broad conclusions are little affected by such potential errors. Given the uncertainties, we assumed a similar level to the one for 2050 in our interim estimate of 2040 economic values.

Agricultural Sector

For the agricultural sector, we focused on the value of crop and livestock production, and agricultural aid. We do not include subsidies, even though these would typically constitute part of government spending and therefore GDP, because it is unclear how subsidies might change under the proposed 30% target. Nor were we able to estimate the global impact on elements of the agricultural economy such as the supply of inputs and labour. We also comment that PA expansion might be expected to increase the level of subsidy in the sector (especially if subsidies follow action and pressure in an increasing number of countries to reward stewardship of environmental public goods as part of a rural support package¹⁶²⁻¹⁶⁵). Any such increase in subsidies would increase the inward flows and agricultural GDP above the estimates we provide.

To calculate the likely changes in post-2030 revenues to the agricultural sector, we input the seven terrestrial scenarios (six for the 30% target plus the non-implementation scenario) into the land-use modules of four Integrated Assessment Models (IAMs): AIM, MAgPIE, IMAGE, and GLOBIOM. These global assessment models are already used to inform policy on topics such as climate change and food security, which they achieve by linking together advanced modelling systems for land use change, global trade and commodity prices, and agricultural productivity⁶⁶. Thus, the IAMs can take a set of constraints such as protected areas and project the likely impact on land use, productivity and prices in the agricultural sector. We focused particularly on croplands and on grass-fed livestock because for agriculture, the main effect of future protected areas will be on the availability of new cropland and grassland, and on the economics of farming on those land use types.

All IAMs use spatially explicit land-use data in half-degree (or higher) resolution as input. Also, all IAMs can provide spatially explicit land-use projections in half-degree resolution as output. For the modelling, however, the spatially explicit layer is aggregated in MAgPIE and downscaled back to half-degree resolution in a post-processing step. In this analysis, we refer to the spatially explicit, half degree resolution layer in the models as “pixel level”. Each pixel can have several land-use types. To project how a 30%-protected-area system would change future agricultural patterns in these models, we coded the rule that protection effectively becomes a subtype of land use, also expressed as a proportion of each pixel that is protected. Therefore, we pre-processed each of the scenarios to re-express it spatially as the proportion of non-productive land in each grid cell that could not be converted to another land-use type. To reflect the likely political timing if the 30% target is agreed at the COP, we implemented half of the protection in 2025, and the remainder in 2030 by linear interpolation, while commenting that faster implementation than this would be preferred by many biologists. Once protection has been implemented in the model’s time flow, the proportion of non-productive land in any pixel is not allowed to fall below the proportion defined by the “protection subtype”. As a result, agricultural expansion sometimes shifts its location, and those locational shifts have knock-on impacts on the expected levels of productivity and farmgate prices. Each model uses a different approach to projecting the four related changes (km² of new agricultural land area, location of new agricultural land area, farmgate price, and agricultural productivity). We used the average outputs across all the models in our main report, but also explored the range of projected values by comparing across the individual model outputs.

The main effect of protected areas is to compel any new agricultural expansion that might have occurred onto the now-protected land to occur elsewhere. In response, farmgate prices and productivity also change, with the precise magnitude of projected change depending on both the scenario studied and the IAM used. Furthermore, reduced land availability (e.g. in IMAGE-MAGNET), also leads to increasing land prices. Price and productivity changes are expressed in the IAMs as indices showing the percentage increase on a 2010 baseline. To estimate the market value of the agricultural sector, we took the 2010 production value of crops and livestock from FAO data¹⁶⁶ and multiplied them by the time, horizon-, scenario- and region-specific price index and productivity index, and then by (one plus) the proportional change in the area of cropland or grassland respectively. We then estimated at a given time horizon and for a given region, for each scenario of increased protected area, the impact of the protected area increases on the market value of the agricultural sector. This was expressed as its gross value. The economic impact of the scenarios on the agricultural sector was calculated as the difference between the market value of the agricultural sector under the 30% scenario and the value under the reference (no increase in protected area coverage) scenario. Furthermore, for AIM and IMAGE, the agricultural values or net output value (production output value minus input

costs) could be computed. We note that AIM models net production output value using a baseline of 2005 FAO data and projects the 2010 values, whereas our calculations of gross output value used FAO 2010 data as the baseline. Where this difference caused a difference in gross values, we used the 2010-baselined version. Net and gross output should not therefore be compared directly in our results.

As with biodiversity, agricultural aid can be considered a form of revenue that flows into the country due to the presence of a sector, contributing to GDP. We took our data on aid to this sector from the OECD summary report of overseas development assistance and other official flows (ODA and OOF) for 2004-2013⁶⁷, where those estimates include imputed flows for aid sources known to be underreported in the OECD aid databases. Agricultural aid has remained relatively flat in recent years (at ~\$12 billion annually), and so we took the assumption that this figure would continue into the foreseeable future. It is difficult to predict long-term changes in likely aid disbursements but for agriculture, the revenue value of aid is such a small percentage of the total revenue achieved from global agriculture (-0.18%) that uncertainty is unlikely to affect the broader conclusions about sectoral revenue.

Forestry Sector

To calculate forestry sector revenues, we inserted the six terrestrial scenarios (plus the reference baseline scenario) into the Global Forest Products Model (GFPM)⁶⁸⁻⁷⁰, applying the same implementation timetable as agriculture in the IAMs and fisheries in the MEMs. The GFPM calculates changes in forestry activity, trade and prices and requires, as input, estimates of the area of exploitable and non-exploitable forest in each of 180 countries⁶⁸⁻⁷⁰. It then outputs the total (gross) value of forestry products from the exploitable portion and the carbon stock in forests, for five- or ten-year time intervals. The calculations include estimates of forest growth (and hence, carbon sequestration) and for the purposes of the current calculation, the model was extended to calculate the changing carbon stocks in non-exploitable (protected) forest as well as exploitable forest. Initial exploitable forest cover was calculated by taking the current protected area maps for each country and overlaying them on the 2015 ESA CCI land use gridded product⁷¹, interpreting land use classes 50-90 as “forest”. Exploitable and non-exploitable (protected) forest cover for the scenarios was calculated similarly. Some classes of protected area (especially IUCN categories V and VI) can allow sustainable use of resources including forestry. We therefore calculated the proportion of category V and VI protected areas in each country currently (based on the WDPA global database at www.protectedplanet.net) and assumed a similar proportion in the post-2020 scenarios, adjusting the exploitable forest area accordingly.

Fisheries Sector

Fisheries revenue in our estimates was primarily composed of the value of the (wild-capture) fishing catch, with additional inputs from fisheries aid, but not including subsidies. The fishing catch is expected to change if marine protected areas (MPAs) are expanded to 30%, but both positive and negative effects are expected. Unlike the more delayed opportunity cost in agriculture, new MPAs can have an instantaneous impact if fleets were already fishing in the area that then becomes protected. Positive effects of protection also occur, however, because MPAs allow the recovery of overfished stocks, which can benefit the industry, but over a potentially longer time period^{14,73,75}. In other words, the negative effects are expected to occur from the outset, whereas the positive effects are expected to occur somewhat later.

To model the impacts of the proposed 30% target on fisheries catch and revenue, we used three Marine Ecosystem Models (MEMs), similar to the IAMs used for terrestrial PAs. MEMs operate in a spatially similar way to IAMs, dividing up the planet into grid cells of either one-degree resolution (for the MEMs BOATS and EcoOcean) or half-degree resolution (for the DBEM model). The scenarios for marine protection were therefore coded as the proportion of each cell that had fishing restrictions. For the two marine scenarios categorized as Biodiversity-Production Compromise (50:50 EEZ and 50:50 Coastal), 50% of the restricted area banned all fishing (“no take”), and 50% allowed sustainable fishing. Sustainable fishing was defined in BOATS as 50% Maximum Sustainable Yield; in EcoOcean as fishing by small-scale fishers only; outputs were not available for DBEM at time of writing (but will become available in Q3 2020). For all other scenarios, all MPAs were 100% no-take. For consistency with the protocol for terrestrial scenarios,

50% of marine protection was implemented in the model in 2025, and the remainder by linear interpolation to 2030. MEMs ran two possible climate forcings: RCP 4.5 and RCP 6.0. Outputs were again provided for ten-year timesteps e.g. 2030, 2040, 2050, up to 2100.

To place a financial value on the catch in each year, BOATS used the Sea Around Us database of landed catch values and applied it to the species mix and weight of fish forecast to be landed in each area and time step. It is possible that changes in demand, effort and international trade would cause the price to vary, but including the kind of general equilibrium trade models that would be needed to adjust landed catch values in response to those dynamics is still a research frontier for ocean ecosystem models, and we had to assume fixed prices. EcoOcean produced projections of catch volumes but not their dollar value, and so we compared change in catch volume across for that model. DBEM results will be reported in later versions of this Working Paper. We note that EcoOcean coded existing MPA coverage as first being implemented in 2020 (owing to the impracticability of re-rendering spatially explicit MPA coverage to one-degree cell equivalents for each historical year available). Although current MPA coverage was largely implemented in recent years⁸, we acknowledge that there was a build-up of coverage over time, and so the immediate post-2020 negative impact on fisheries catch will be an overestimate because it reflects an artefactual shock.

We were unable to model the impact of new protected areas on a possible expansion of shrimp fisheries at the expense of felling coastal mangrove forests. To do so would have required a detailed spatial model of where shrimp farming might expect to go under business as usual, of where it would alternatively go if some mangrove areas were protected, and of the impact that geographic displacement of the productive areas would have had on revenues and costs, none of which is parameterizable in the global models. We also omitted possible aquaculture effects for a similar reason. Nor was it possible in this study to predict how patterns of global shipping might alter (with cost implications) in response to MPA expansion.

We included fisheries international aid flows as revenue, for the same reasons already specified for the nature sector and agriculture sector. We used Blasiak et al. 2017¹⁶⁸ as our data source. That paper specifies that over recent years (2010-2015), global fisheries aid has fallen by over 30% in value, to only \$0.166 billion dollars. To reflect the ongoing decline, we arbitrarily assumed that by 2050, aid would be 50% of its current value. As with agriculture, the overall expected revenue from fisheries today is approximately 1000 times greater than this, and so any inaccuracy in estimating future fisheries aid flows is unlikely to affect overall conclusions about the sector. However, we comment that under the 30% proposal, fisheries aid and subsidies would be expected to increase dramatically to temporarily compensate local fishing communities for revenue losses while stocks recover. However, it is difficult to predict how much compensatory support might flow into the industry to assist in this transition to sustainability and, not wishing to make an arbitrary estimate, we simply left this figure as unknowable. Fishers may therefore experience only a small portion of the short-term economic losses that we estimate.

Avoided-cost and Ecosystem Service Values

A further important value of nature protection was avoided cost. In its broader sense, avoided cost is equivalent to the value of the ecosystem services (ES) that would have been lost if an area was not protected (i.e. a counterfactual, measuring the difference between the ES under each 30% scenario and the ES under the no-PA-expansion scenario). Avoided ES costs vary strongly depending on where the protection is conferred, a fact that would impact the expected results from our scenarios of where protection might be located. Assessing spatial variation in ES costs typically requires a meta-analytic function that takes all known individual studies on ES in a biome, and then fits a metaregression model based on spatially explicit predictors to them (for example, local human GDP per capita or population density¹⁶⁹⁻¹⁷¹). The proposal to expand protected areas globally will likely affect all biomes, but only a subset of biomes have metaregression functions that would allow the calculation of spatially explicit estimates. Following the examples of The Economics of Ecosystems and Biodiversity global study (TEEB)¹⁷¹ and Brander et al. 2020¹⁴, we estimated avoided cost functions only for better-studied ecosystems, with the secondary proviso that any biome also needed to have a clear relationship to the land-use classes in the IAMs or the parameters of the MEMs.

On land, the main biome that satisfied these two conditions was tropical forest, which has a global meta-analytic value function estimated in Carrasco et al. 2014³⁴. We took the spatially explicit tropical forest valuation layer from that study and, for each timestep in each terrestrial scenario in each IAM, applied the function to the total forest cover in each grid cell. We then estimated the value of each scenario for 2050 and 2040, as the difference between the 30%-PAs value and the counterfactual no-PA-expansion value. The change itself was calculated relative to a 2030 baseline, this being the moment when all the protection is designed to be in place under the 30% proposal.

The TEEB assessment does contain a tentative grassland category but as that assessment itself stated, the value function is very low-confidence⁷¹. To examine whether grassland change might be of sufficient magnitude to nevertheless justify an ES value estimate with all the uncertainties involved, we first used the IAM outputs to calculate the differences in the extent of the (managed) grassland land-use class in 2050 under the 30%-PAs scenarios, as compared against the counterfactual of the no-PA-expansion scenario. The difference was generally less than 1%, so we judged that fitting a highly uncertain grassland function to them would introduce more error than information and did not attempt to calculate spatially explicit grassland shadow values. Unmanaged grassland is not separately defined as a separate land-use class in the IAMs, being merged with e.g. tundra and rock ecosystems in the “Other” category, confirming our decision.

Our estimates of avoided costs on land therefore omit the value of protecting any other terrestrial ecosystems e.g. boreal or temperate forest, tundra, or savannah, and also any value was acquired in the process of implementing new protected areas between a COP decision and 2030.

For the marine aspect of the proposed 30% target, two biomes could satisfy the dual inclusion condition (a global metaregression function and a clear relationship to the outputs from the MEMs or IAMs): mangroves and coral reefs. (We could not project impacts of the 30% target on land use change in wetland areas, despite the economic importance of this biome^{14,172-174}). For mangrove value, we used a similar approach to the one for tropical forests: we estimated the forecast change in mangrove cover per country for 2030-2040 and 2030-2050 in each scenario including the no-expansion baseline, then applied the spatially explicit metaregression function to calculate the differenced net present value of only the specific mangrove areas that would be lost without protection.

For coral reefs, the main ecosystem services included recreational diving, snorkelling and fishing, tourism more broadly, benefits to fisheries, and coastal protection⁷¹. However, our revenue calculations already included large financial values for all the visitor activities broadly associated with reefs (including diving and snorkelling) and our fisheries models already assessed benefits of protection to commercial fisheries, so applying the metaregression function would clearly have constituted repeated double counting. The main shadow value not yet included for reefs is the degree to which expanded protection of reefs would reduce coastal damage by 2050, compared to a counterfactual of the current protected area system. But since coastal protection is provided by the reef substrate and this is unlikely to disappear in its entirety by 2050, the key counterfactual question is how far protection would slow rates of reef degradation and its impact on coastal damage during 2030-2050¹⁷⁵, bearing in mind that not all reefs would be degraded in the absence of protection, not all unprotected reefs would be automatically degraded, and that the diving and boating attracted by protected coral reefs can itself cause significant coral damage¹⁷⁶. It was not possible to extract a separate metaregression function that only considered this service and these complexities, but we considered that the bias introduced by multiple double counting would be considerably more serious than the bias introduced by not accounting for how coral diversity might degrade more slowly than against counterfactual in some expanded MPAs and mitigate coastal damage from wave action. We therefore judged it best to include only the tourism, recreation and fisheries revenues in our reef values, noting that this generates an underestimate of the true value of protecting more reefs under the 30% proposal.

For carbon values, we followed previous studies in extracting these separately from the metaregression functions^{14,34,171}. Our carbon calculations were based on the outputs from the Global Forest Products Model (GFPM, see Forestry section for description), and therefore refer only to tree cover, and not to possible carbon changes when unprotected non-tree-covered land is protected. The GFPM accounts for biomass growth and its relationship to exploitation in the non-protected area of forest in each country, as well as simple biomass loss from felling. Our

carbon calculations measure the difference between total biomass under the no-PA-expansion baseline and total biomass in the 30% scenarios, at each time point (2020-2070, in ten-year intervals). To assign a value to the carbon difference in each timestep, we took the “central” social cost of carbon values for achieving a 2.5 degrees Celsius or less temperature increase from Nordhaus et al. 2017¹⁷⁷ (using the “maximum for 100 yrs” option to be conservative, since this gives the lower price). We could not reject a hypothesis of linear change in biomass between 2030 and 2050, so we assumed linear change and calculated the difference in difference of national carbon stocks for 2030-2050. (Specifically, we calculated the difference between the 2030-2050 change for the baseline no-PA-expansion scenario, and the same change for each of the 30% scenarios, then multiplied this by the value per tCO₂eq for the year in which the difference in difference was being calculated). In annualising these values, we assumed that the value of a decade’s avoided emissions would be paid out in equal instalments over the next 10 years, assuming an inflation rate of 3%.

A Note on Discounting Versus Forward Planning

The current study focuses on the total multi-sector revenues expected under different policy options and the associated investment/costs required to capture those revenues. The comparison of interest is between the annual revenues that protected area systems will capture in the future, and the annual costs associated with capturing those revenues. In such a case, there are two options. The first and simplest option is to directly compare the revenues and costs projected in future years. If a policy action yields greater benefits in a future year than not taking action, a public investment case exists. The question arises whether those future benefits and the costs should be discounted^{178,179}, on the grounds that future benefits are less valuable than present benefits, or future costs less of a burden than present costs. We assume that the costs will largely be met out of public spending budgets and for those, both discounting and the precise choice of discounting rate are controversial^{178,179}. Moreover, discounting would only affect our conclusion of a net positive future-years outcome if the costs were rising faster than the benefits, or if costs were discounted at a different rate from the benefits. In our calculations, benefits were increasing faster than expected costs and also faster than the rate of inflation, and there was no *a priori* reason to make the controversial assumption that costs and benefits should have different discounting rates. We therefore chose the simpler first option and present the constant-dollar value in each future year of the costs and benefits. (Estimated present values of the different costs and benefits can be calculated for any time period required, and parties should approach the authors if these are of interest.)

COSTS/INVESTMENT NEEDS

The most commonly used algorithms for calculating budget needs in protected areas are those of Balmford, Bruner and colleagues¹⁸⁰⁻¹⁸². These are based on the operating cost requirements stated by the managers of individual protected areas approximately 20 years ago. However, at the national level where most budgeting decisions for PA systems are taken (and from where most funds to protected area systems still originate), there are a number of other costs involved, including the central costs of the coordinating protected-area authority, capital investment costs, and even community outreach beyond PA boundaries. Moreover, expert PA cost assessments may have changed in the past 20 years, not least because of the increased pressure of human activity on protected areas^{133,183,184}. We therefore began by updating cost information on minimum adequate budgets, using the decision-making scale of national PA networks (as done by James et al.^{117,185}).

Politically, bottom-up approaches to costing can seem preferable, because the information comes from the decision-makers who will implement any changes. In recent years, many governments have reported their own bottom-up assessments of minimum adequate budget needs for their protected area systems. In particular, the United Nations Development Programme (UNDP) has collated a number of standardized reports from countries associated with recent Global Environment Facility (GEF) rounds of funding. A number of other governments have also carried out their own studies, even without the mandate of the GEF, and in other cases, an independent academic or NGO carried out the study. We therefore worked with the UNDP to analyse the data from the GEF-motivated reports, known as the “Financial Scorecards”, and supplemented these with a further multi-language search for individual government and independent reports on national budget needs. In total, we managed to find 97 countries that had reported the budget needs for their national PA system.

Our aims were to generate global budget need estimates for the current protected area system, and for the various possible expansions of the system represented by our scenarios. We therefore built a statistical model of the known budget needs, by generating hypotheses about drivers of need, assembling data on those drivers for all countries, and testing which combinations of factors explained the data better.

The financial scorecards themselves suggest that marine protected areas may have a different cost structure, implying that national systems that are strongly marine-biased may affect the model. We therefore split the analysis into two parts: the first “terrestrial/mixed” part excludes all national systems with a marine component greater than 55% of the whole (by area); and the second “marine/coastal-dominated” part analyses separately the countries where marine and coastal PAs predominated. (Sensitivity testing suggested that varying the 55% cut-off did not affect the modelling outcomes substantively). The response we modelled was the standard measure of cost per hectare¹⁸⁰⁻¹⁸².

Terrestrial Protected Areas: Annual Budget Needs

Following previous researchers, the variables we theorized to influence the budget needs were the local cost base, the degree of human pressure that the PA system has to resist, and economies of scale^{180-182,186}. Following Waldron et al.^{133,187}, we corrected for national differences in this cost base, we used purchasing power parity¹⁸⁸, multiplying the cost per hectare by the national price index. For pressure, a common metric is the Human Footprint^{183,184,189}, which could be regarded as a form of supply-side pressure. On the demand side, the potential production value of the land or its resources is also likely to increase the pressure. The value of potentially productive land is commonly expressed as agricultural rent^{170,190} and in ocean contexts, the most significant productive value is the fisheries catch that could be achieved. If a country places most of its PAs in areas of prime agricultural land or dense human populations, there will be high pressure on the PA and also a higher cost base in the region (because, for example, land with naturally high agricultural productivity has high rent values, putting upward pressure on local consumer-goods prices and wage expectations, and resulting in a generally more expensive place to live and operate).

Typically, it is the raw value of pressure metrics that is used in statistical analysis. However, the likely influence of agricultural rent and the human footprint on PA costs has two important nuances. In a country with generally low agricultural productivity, it is not the absolute value of agricultural rent that is important to the population, but the value relative to all the other locations where farming might take place. For example, an arid country would typically have low rent values everywhere, but it would be misleading to believe that because of that, there was no pressure on any of its natural land. Instead, the pressure will fall on the areas that are most productive relative to the national mean. The same applies to the Human Footprint: in countries where human pressure is low everywhere, national-scale costs could still be higher in the places with the densest populations and most intense economic activity, even if those values are low in absolute terms. We therefore re-expressed both agricultural rent and the Human Footprint as relative to the national mean, or formally, mean PA value/mean national value.

The values of agricultural rent and human footprint in the buffers may be a theoretically better predictor of cost than the values inside PAs themselves. This could occur if rent inside the PA is known but its likely value is signalled by rent just outside the PA; or if farming is drawn to areas of relatively high rent and, if those are on the borders of PAs, then there will be pressure to expand across those borders. We therefore extracted rent, footprint, relative rent and relative footprint values for both PAs and for a 25km buffer surrounding each PA (but excluding water). To convert up to national levels (the scale of the response variable), we took the mean value of each variable for all PAs/PA buffers in each country. For the Human Footprint, we used the dataset of Venter et al., updated to reflect recent improvements in the data¹⁸⁴. For the agricultural rent predictor, we used the dataset of Carrasco et al.¹⁷⁰, again with updates by the authors.

Economies of scale can apply both to individual PAs and to national PA systems. At the local scale, a PA of ten hectares can be patrolled by the same guard force as a PA of five hectares, making the budget per hectare cheaper (local economy of scale). At the national scale, individual PAs all share the same central administration force and can even share some mobile equipment or human resources, so a larger PA system would also be expected to cost less per hectare than a small one. We therefore calculated three possible metrics of PA size: (i) the mean area and

(ii) mean log-transformed area for all the individuals PAs in each national system at the date corresponding to the budget need assessment; and (iii) the total national area referenced in the assessment, also measured for the year in which the assessment was carried out. All three metrics were strongly co-correlated, so we tested them separately and used model selection to determine which explained more of the data.

Governance strength is also likely to influence PA costs, although in this case, the expected effect is not theoretically clear. On the one hand, countries with high governance scores are typically wealthy, and so there may be a lower tendency to put pressure on a legally registered PA, even if it seems economically inviting to do so. On the other hand, countries with higher GDP tend to have both higher costs and stronger governance¹⁸⁷, potentially causing a positive correlation between costs and governance scores. We therefore tested for the predictive power of governance without a prior expectation of the sign of the relationship, and in a separate set of tests, for the predictive power of the GDP per capita (in \$PPP). For the governance metric, we used the Government Effectiveness score from World Bank Groups' Worldwide Governance Indicators dataset¹⁹¹, since this has been associated with biodiversity finance patterns in the past^{187,192}. GDP per capita was taken from World Bank data, using the mean of 2012-2017 to reflect the period over which the majority of the bottom-up cost estimates were made. One might expect governance to reduce the likelihood that an economically attractive PA will actually be invaded. We therefore tested for interactions with agricultural rent and footprint, for both governance and its co-correlate GDP.

A further potential predictor of budget need is the relative amount of the system that is being strictly protected (compared to allowing human uses). We therefore created a dummy variable for the proportion of the system that was IUCN category V or VI (indicating that sustainable human use, mostly envisaging resource extraction, is allowed).

All of the pressure predictors treat PAs as if they were parts of nature distinct from human economic activity, and therefore prone to being invaded by that activity. However, as emphasized in the Main Report, PAs in the 21st century are very much revenue-generating parts of the national economy. They therefore function as businesses with variable costs, in which case one might expect higher business volumes (higher revenues) to be associated with higher costs. The UNDP Financial Scorecards and many of the independent reports also quantify the level of revenue that is generated by national PA systems (where that revenue is most commonly associated with tourist visits and associated revenue streams such as concessions, retail, boating or camping fees, etc). We therefore tested whether revenues predicted costs, expressing revenues on a per-hectare \$PPP basis, for consistency with the cost metric.

Finally, marine PA costs may well be different from terrestrial PA costs, but many national PA systems reported their budget needs for a composite of marine and terrestrial PAs. Since this different ratio of TPA-to-MPA should affect the national budget need, we included the ratio as a potential predictor.

After selecting and collecting data for all the above variables, we created statistical models, each containing a subset of variables, to test for their ability to predict the observed data. Budget needs cannot take negative values, so we used generalized linear modelling (GLMs) as our statistical framework. We initially tested a Poisson error structure but this generated strong overdispersion, so in our final modelling approach we used a Negative Binomial error structure. For model comparison, we used an Information Theoretic Approach, calculating the Akaike Information Criterion values of each model with correction for small sample sizes (AICc values) and regarding all models with delta AICc <2 as the best-fit model set. Prior to analysis, we log transformed the "areal extent of the national PA system" variable and the revenue variable. The dependent variable itself (the budget need per hectare) also had a strongly skewed statistical distribution, so we tested separate sets of models with a square root transformation (since the budget need is a count variable) and a natural log transformation. The square root transformation generated the best fit to data and is reported in the final results.

The best-fit model set contained two models. In the model with the lowest AICc (pseudo R squared = 0.9), budget need per hectare decreased as the total size of the protected area system increased. PA systems with a higher mean net agricultural rent in the buffer, relative to the mean agricultural rent of the country, had higher budget needs. Similarly, buffer footprint relative to the national mean generally increased budget need, but in interaction with the governance term. Management cost was positively predicted by the level of revenue and this was the largest effect size. Finally, the percentage of the assessed system that was marine improved the fit (AICc), although the nature of the

effect was unclear since the confidence bounds of the parameter estimate overlapped zero. The other best-fit model was similar but omitted the term for the percentage of marine PAs in the system and explained 85% of the variation.

We therefore applied the model outputs to predict the likely costs of the six terrestrial systems represented by the 30%-protection scenarios. An expanded terrestrial system could be associated with a wide range of possible marine system expansions, so to simplify and universalise forecasting, we applied the best-fit model that did not contain the percentage-marine term. Details of the model coefficients for this and other costing analyses are available from the authors upon reasonable request.

Marine Protected Areas: Annual Budget Needs

We used a similar method to estimate the investment needs for each marine scenario. We compiled a dataset of where countries had estimated the budget needs to adequately manage their entire national MPA networks (n=47) and created statistical predictive models using expected budget-need drivers. For pressure variables for marine PAs, we first calculated the weight of the fisheries catch in the immediate 25km ocean area around the MPA. The metric of the relative rather than absolute pressure was calculated by dividing catch weight in the buffer by mean catch weight across the entire EEZ (Exclusive Economic Zone). The impact of coastal population density and GDP has been explored in past predictive models of individual MPA costs, as has the distance offshore^{182,193}. However, we theorized that the further offshore an MPA was, the less important the coastal socioeconomic characteristics would be. We therefore measured the population density and GDP per capita (2012-2017 mean) of all coastal populations at the point on the coast nearest the MPA, up to a standardized distance inland (5km), and then modelled possible interactions between the MPA's distance offshore and those socioeconomic factors. Finally, we measured the size of both individual MPAs and of the entire national MPA system, to assess whether economies of scale reduce costs per hectare (similarly to the TPA example).

We took mean 2011-2015 catch values from data collated by Watson et al^{194,195}, and extracted the human population density and GDP per capita data by overlaying the Gridded Population of the World and gridded GDP per Capita spatial datasets^{196,197} on the portions of coastline closest to the MPAs.

For statistical analysis, the budget need, relative catch weight and MPA system area had skewed frequency distributions and were log-transformed. As with the terrestrial budget need analysis, we then used a Generalized Linear Model with a negative binomial error structure to model the investment needed per hectare in national marine PA systems, and an information theoretic framework to select the best-fitting models. The best-fit model used relative catch weight, coastal GDP per capita and the MPA system size as the final predictor variables. The second-best model (delta AICc = 10.06) was identical except that it dropped the relative catch weight. The model with catch weight explained 90.4% of the variation in the data, and the second-best model explained 87% of the variation. The final sample size for analysis was 43 because we could not calculate all predictor variables for four countries with budget need data. As expected, the budget need per hectare went down with larger systems, and higher when coastal GDP was higher (which we interpret as reflecting both a higher local cost base and higher local pressure). It also increased as the nationally scaled catch in the MPA buffers increased, which we again interpret as creating a situation where the value of the resources in the MPA is higher and so it requires more effort to defend them. We tested for a number of possible interactions, but none was selected in the two models with lowest AICc values.

We applied the model to predict the budget needs for expanding the MPA system, calculating spatially explicit values for the model parameters under each scenario and inputting these as the dataset for the model to predict from. We were not able to forecast future values of pressure variables such as the Human Footprint because these rely on spatially explicit mapping of road networks (among other socioeconomic indicators) that could not be projected into the future, and so we had to use current values instead. Owing to difficulties in calculating relative catch when the spatial location of new MPAs is not well defined, we used also the second marine model (without relative catch) to predict MPA costs. However, we note that the coefficients for GDP per capita and MPA area are both lower in the best-fit model than in the model applied. The economy of scale will therefore be overestimated (since its coefficient is negative) but the influence of pressure will be underestimated. When these effects are combined, the second-best model seemed likely to overestimate costs for the marine component.

Establishment Costs

The main site components of establishment cost are the need for new infrastructure and the cost of land purchase. There are also administrative costs involved but these are very little studied^{181,198}, making it difficult to include them in any global assessment of a future PA system. Our costing models include minimum adequate capital spending needs (CAPEX, including infrastructure), again derived from countries' own estimates. However, they were unable to account for possible differences between CAPEX needs in existing and newly-established PAs. This may cause an underestimate of CAPEX needs, although a large number of countries stated that their existing CAPEX budgets were already very low or even zero, so the magnitude of underestimation may be quite small.

Land purchase costs are difficult to estimate because there is no global database of land prices and indeed, many PAs have been established on land for which purchase is inappropriate e.g. government-owned land or land with communal ownership by an IPLC group. A common proxy for agricultural land price is the net farm income divided by the capitalization rate of agricultural land (where income has to be modelled based on the expected production and costs of not-yet-converted land)¹⁰⁵. We calculated this proxy for all the new land in each scenario, noting that this will give an overestimate, since it ignores both yield gaps^{106,107} and cost savings on government or indigenous lands. Conversely, the capitalization rates, taken from European farms¹⁰⁵, are likely to also overestimate costs in non-EU countries. Thus, CAPEX is underestimated but purchase cost is overestimated, partially cancelling out much of their respective errors in the calculation of establishment cost. The cost of a large land purchase for a long-term protected area program would typically be subject to some form of loan or national debt arrangement, rather than coming out of the first year's budget in a single payment. We therefore annualised the land purchase cost by amortising it over 20 years.

For marine protected areas, the most recent estimate is a creation cost of \$42860 per MPA, irrespective of the size or location of the MPA but rising if creation takes longer than average¹⁹³. To apply this to our marine scenarios, we therefore needed an estimate of how many new MPAs would be created in each country under each scenario. Part of the expansion to 30% would be achieved by expanding existing areas rather than creating new ones, so to estimate the number of new MPAs that might be needed, we assumed that existing MPAs would double in size, creating a partial expansion, and then used the number of MPAs in that partial expansion to estimate the number of MPAs that would be needed to complete the expansion to 30% in each EEZ. We note that the annualised amortised establishment costs for new MPAs amount to less than one percent of the annual management cost, and so uncertainties in this particular cost component have a minimal impact on the overall budget need estimate.

Current Global Spending on Protected Areas

We collated current budget allocations to national protected area networks using a similar protocol to the one used for collating budget needs. We were able to source primary estimates for 124 countries. The data-deficient countries and territories were mostly small offshore island territories, microstates, small island states and lower-income countries, all of which are unlikely to have budgets exceeding a few million dollars. The main exceptions were Venezuela (although its recent PA budget is not likely to be large), Portugal, Japan, Iran, Finland and Switzerland. An additional important consideration was that much of the European Union budget information has not been updated for ten years, during which time the protected area estate (and so presumably the budget) has increased. We corrected for this by multiplying the per-hectare spend in the 2009 EU data by the additional hectares added since 2009, taking the extents of national PA systems from the World Database of Protected Areas (www.protectedplanet.net). The global total for spending on protected areas from these data sources was \$17.2 billion; we then imputed values for missing countries by using data from comparable countries and adjusting for PA network size (imputed additional spending = \$2.5 billion). An additional \$4.5 billion was added as the mean contribution calculated from local communities to PA management costs¹¹⁵. Specifically for MPAs, a further 11 countries had no data but may have had a non-trivial budget (Vanuatu, Syria, Mauritania, Malta, Morocco, Monaco, Libya, Lebanon, Haiti, Barbados, Antigua). However, we judged it unlikely that these missing data points would sum to more than a few million dollars per year. Therefore, our overall estimate for global PA spending was approximately \$24.3 billion. We note that the spending subtotal in the MPA-focused (35 countries) was only \$282 million, although many countries in the broader 'mixed' analysis will have included MPA spending as part of their total spending, and so the marine-specific budget cannot be exactly calculated.

REFERENCES

1. IPBES. *The global assessment report on biodiversity and ecosystem services. Summary for policymakers.* (2019).
2. World Economic Forum. *The Global Risks Report 2020.* http://www3.weforum.org/docs/WEF_Global_Risk_Report_2020.pdf (2020).
3. Butchart, S. H. M. *et al.* Protecting important sites for biodiversity contributes to meeting global conservation targets. *PLoS One* **7**, (2012).
4. Rodrigues, A. S. L. *et al.* Global Gap Analysis: Priority Regions for Expanding the Global Protected-Area Network. *Bioscience* **54**, 1092-1100 (2004).
5. Chape, S., Harrison, J., Spalding, M. & Lysenko, I. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. *Philos. Trans. R. Soc. B Biol. Sci.* **360**, 443-455 (2005).
6. Gray, C. L. *et al.* Local biodiversity is higher inside than outside terrestrial protected areas worldwide. *Nat. Commun.* **7**, 12306 (2016).
7. UNEP-WCMC, IUCN & NGS. *Protected Planet Report 2018.* https://livereport.protectedplanet.net/pdf/Protected_Planet_Report_2018.pdf (2018).
8. Marine Conservation Institute. MPA Atlas. www.mpatlas.org.
9. *Global Environment Outlook - GEO - 6.* (2019).
10. Dinerstein, E. *et al.* A Global Deal for Nature: Guiding principles, milestones, and targets. *Sci. Adv.* **5**, (2019).
11. O'Leary, B. C. *et al.* Effective Coverage Targets for Ocean Protection. *Conserv. Lett.* **9**, 398-404 (2016).
12. Congress, W. P. A strategy of innovative approaches and recommendations to enhance implementation of marine conservation in the next decade. (2014).
13. CBD. Zero Draft of post-2020 biodiversity framework. *Secr. Conv. Biol. Divers.* (2020).
14. Brander, L. M. *et al.* The global costs and benefits of expanding Marine Protected Areas. *Mar. Policy* **116**, 103953 (2020).
15. Figgis, E. P. *et al.* *Valuing Nature Protected Areas and Ecosystem Services.* (2015).
16. Oldekop, J. A., Holmes, G., Harris, W. E. & Evans, K. L. A global assessment of the social and conservation outcomes of protected areas. *Conserv. Biol.* **30**, 133-141 (2016).
17. Stolton, S. *et al.* Values and benefits of protected areas. in *Protected area governance and management* (eds. Worboys, G. L., Lockwood, M., Kothari, A., IFeary, S. & Pulsford, I.) 145-168 (ANU Press, 2015).
18. Dudley, N. *et al.* The essential role of other effective area-based conservation measures in achieving big bold conservation targets. *Glob. Ecol. Conserv.* **15**, 1-7 (2018).
19. Jonas, H. D. *et al.* Will 'other effective area-based conservation measures' increase recognition and support for ICCAs? *Parks* **23**, 63-78 (2017).
20. World Travel & Tourism Council. *The Comparative Economic Impact of Travel & Tourism.* http://www.wttc.org/-/media/files/reports/benchmark_reports/the_comparative_economic_impact_of_travel__tourism.pdf (2012) doi:10.1074/jbc.M411161200.

21. UK Department for International Development. *Agriculture and Growth*. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/321107/Agri-growth1.pdf (2014).
22. Dzemydaite, G. Agriculture'S Impact for the Economy: Inter-Industry Linkages and Multiplier Effects. in *Proceedings of the 8th International Scientific Conference Rural Development 2017* (2018). doi:10.15544/rd.2017.057.
23. Dudley, N. & Stolton, S. *Running pure: the importance of forest protectd areas to drinking water*. (2003).
24. Dyck, A. J. & Sumaila, U. R. Economic impact of ocean fish populations in the global fishery. *J. Bioeconomics* **12**, 227-243 (2010).
25. Moody's. Green, social and sustainability bond issuance to jump 24% in 2020 to \$400 billion. (2020).
26. Climate Bonds Initiative. 2019 Green Bond Market Summary. (2019).
27. Nepal, P., Ince, P. J., Skog, K. E. & Chang, S. J. Forest carbon benefits, costs and leakage effects of carbon reserve scenarios in the United States. *J. For. Econ.* **19**, 286-306 (2013).
28. Nepal, P., Ince, P. J., Skog, K. E. & Chang, S. J. Projected US timber and primary forest product market impacts of climate change mitigation through timber set-asides. *Can. J. For. Res.* **43**, 245-255 (2013).
29. Lambin, E. F. & Meyfroidt, P. Global land use change, economic globalization, and the looming land scarcity. *Proc Natl Acad Sci U S A* **108**, 3465-3472 (2011).
30. UN Department of Economic and Social Affairs. *2018 Revision of World Urbanization Prospects*. (2018).
31. Leimbach, M., Kriegler, E., Roming, N. & Schwanitz, J. Future growth patterns of world regions – A GDP scenario approach. *Glob. Environ. Chang.* **42**, 215-225 (2017).
32. Sala, E. *et al.* A general business model for marine reserves. *PLoS One* **8**, e58799–e58799 (2013).
33. McCook, L. J. *et al.* Adaptive management of the Great Barrier Reef: A globally significant demonstration of the benefits of networks of marine reserves. *Proc. Natl. Acad. Sci.* **107**, 18278 LP – 18285 (2010).
34. Carrasco, L. R., Nghiem, T. P. L., Sunderland, T. & Koh, L. P. Economic valuation of ecosystem services fails to capture biodiversity value of tropical forests. *Biol. Conserv.* **178**, 163-170 (2014).
35. Allen, T. *et al.* Global hotspots and correlates of emerging zoonotic diseases. *Nat. Commun.* **8**, 1124 (2017).
36. MacKinnon, K., van Ham, C., Reilly, K. & Hopkins, J. Nature-Based Solutions and Protected Areas to Improve Urban Biodiversity and Health BT - Biodiversity and Health in the Face of Climate Change. in (eds. Marselle, M. R., Stadler, J., Korn, H., Irvine, K. N. & Bonn, A.) 363-380 (Springer International Publishing, 2019). doi:10.1007/978-3-030-02318-8_16.
37. UN. *Transforming our world: the 2030 agenda for sustainable development*. (2015).
38. Garnett, S. T. *et al.* A spatial overview of the global importance of Indigenous lands for conservation. *Nat. Sustain.* (2018).
39. Rockström, J. *et al.* A safe operating space for humanity. *Nature* **461**, 472-475 (2009).
40. Díaz, S. *et al.* Pervasive human-driven decline of life on Earth points to the need for transformative change. *Science (80-.)*. **366**, eaax3100 (2019).
41. UNEP. *Global Environment Outlook 6*. (2019).
42. Di Minin, E. & Toivonen, T. Global Protected Area Expansion: Creating More than Paper Parks. *Bioscience* **65**, 637-638 (2015).
43. World Travel and Tourism Council. *The economic impact of global wildlife tourism - Travel and tourism as an economic tool for the protection of wildlife*. <https://travesiasdigital.com/wp-content/uploads/2019/08/The-Economic-Impact-of-Global-Wildlife-Tourism-Final-19.pdf> (2019).

44. Balmford, A. *et al.* Walk on the Wild Side : Estimating the Global Magnitude of Visits to Protected Areas. 1-6 (2015) doi:10.1371/journal.pbio.1002074.
45. World Tourism Organization. *Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa – Briefing Paper. Towards Measuring the Economic Value of Wildlife Watching Tourism in Africa – Briefing Paper* (2014) doi:10.18111/9789284416752.
46. Giants, S. for, Capital, C. & UNEP. *Building a wildlife economy. Working paper 1: developing nature-based tourism in Africa's State Protected Areas.* <https://spaceforgiants.org/wp-content/uploads/2019/06/Building-Africas-Wildlife-Economy-Space-for-Giants-Working-Paper-1.pdf> <http://www.pnas.org/lookup/doi/10.1073/pnas.1704949114> (2019) doi:10.1073/pnas.1704949114.
47. Cullinane Thomas, C., Koontz, L. & Cornachione, E. *2018 National Park visitor spending effects.* (2019).
48. Sathirathai, S. & Barbier, E. B. Valuing mangrove conservation in Southern Thailand. *Contemp. Econ. Policy* **19**, 109-122 (2001).
49. Ferraro, P. J. *et al.* Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation. *Proc. Natl. Acad. Sci.* **112**, 7420-7425 (2015).
50. Krieger, D. *The economic value of forest ecosystem services: a review.* vol. 6 (2001).
51. Naidoo, R. *et al.* Evaluating the impacts of protected areas on human well-being across the developing world. *Sci. Adv.* **5**, eaav3006 (2019).
52. Hansen, M. M., Jones, R. & Tocchini, K. Shinrin-yoku (Forest bathing) and nature therapy: A state-of-the-art review. *Int. J. Environ. Res. Public Health* **14**, (2017).
53. Alpizar, F. & Bovarnick, A. *Targeted Scenario Analysis. A new approach to capturing and presenting ecosystem service values for decision making.* (2013).
54. Wenhai, L. *et al.* Successful Blue Economy Examples With an Emphasis on International Perspectives. *Frontiers in Marine Science* vol. 6 261 (2019).
55. Miller, D. J. & Plantinga, A. J. Modeling Land Use Decisions with Aggregate Data. *Am. J. Agric. Econ.* **81**, 180-194 (1999).
56. Plantinga, A. J. The Effect of Agricultural Policies on Land Use and Environmental Quality. *Am. J. Agric. Econ.* **78**, 1082-1091 (1996).
57. Verburg, P. H., Mertz, O., Erb, K. H., Haberl, H. & Wu, W. Land system change and food security: Towards multi-scale land system solutions. *Curr. Opin. Environ. Sustain.* **5**, 494-502 (2013).
58. Wu, J. & Segerson, K. The Impact of Policies and Land Characteristics on Potential Groundwater Pollution in Wisconsin. *Am. J. Agric. Econ.* **77**, 1033-1047 (1995).
59. Chakir, R. & Lungarska, A. Agricultural rent in land-use models: comparison of frequently used proxies. *Spat. Econ. Anal.* **12**, 279-303 (2017).
60. Lichtenberg, E. Land Quality, Irrigation Development, and Cropping Patterns in the Northern High Plains. *Am. J. Agric. Econ.* **71**, 187-194 (1989).
61. Balmford, A., Beresford, J., Green, J., Naidoo, R. & Walpole, M. *A Global Perspective on Trends in Nature-Based Tourism.* **7**, 1-6 (2009).
62. Bosetti, V., Lubowski, R., Golub, A. & Marakndya, A. Linking reduced deforestation and a global carbon market: implications for clean energy technology and policy flexibility. *Environ. Dev. Econ.* **16**, 479-505 (2011).
63. Barbier, E. B., Burgess, J. C. & Dean, T. J. How to pay for saving biodiversity. *Science (80-.)*. **360**, 486 LP - 488 (2018).
64. ten Kate, K., Bishop, J. & Bayon, R. *Biodiversity offsets: views, experience, and the business case.* (2004).

65. Koh, N. S., Hahn, T. & Boonstra, W. J. How much of a market is involved in a biodiversity offset? A typology of biodiversity offset policies. *J. Environ. Manage.* **232**, 679–691 (2019).
66. Popp, A. *et al.* Land-use futures in the shared socio-economic pathways. *Glob. Environ. Chang.* **42**, 331–345 (2017).
67. Stehfest, E. *et al.* Key determinants of global land-use projections. *Nat. Commun.* **10**, 2166 (2019).
68. Buongiorno, J., Zhu, S., Zhang, D., Tomberlin, D. & Turner, J. *The Global Forest Products Model: Structure, Estimation, and Applications.* (Academic Press, 2003).
69. Buongiorno, J. Global modelling to predict timber production and prices: the GFPM approach. *For. An Int. J. For. Res.* **88**, 291–303 (2014).
70. Nepal, P., Korhonen, J., Prestemon, J. P. & Cubbage, F. W. Projecting global planted forest area developments and the associated impacts on global forest product markets. *J. Environ. Manage.* **240**, 421–430 (2019).
71. ESA. Land Cover CCI Product User Guide Version 2. Tech. Rep. (2017).
72. Duarte, C. M. *et al.* Rebuilding marine life. *Nature* **580**, 39–51 (2020).
73. Davis, K. J., Vianna, G. M. S., Meeuwig, J. J., Meekan, M. G. & Pannell, D. J. Estimating the economic benefits and costs of highly-protected marine protected areas. *Ecosphere* **10**, (2019).
74. Buxton, C. D., Hartmann, K., Kearney, R. & Gardner, C. When is spillover from marine reserves likely to benefit fisheries? *PLoS One* **9**, 1–7 (2014).
75. Kerwath, S. E., Winker, H., Götz, A. & Attwood, C. G. Marine protected area improves yield without disadvantaging fishers. *Nat. Commun.* **4**, 2347 (2013).
76. Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P. & Goodridge, R. Effects of marine reserves on adjacent fisheries. *Science (80-.)*. **294**, 1920–1923 (2001).
77. Lotze, H. K. *et al.* Global ensemble projections reveal trophic amplification of ocean biomass declines with climate change. *Proc. Natl. Acad. Sci.* **116**, 12907 LP – 12912 (2019).
78. Galbraith, E. D., Carozza, D. A. & Bianchi, D. A coupled human-Earth model perspective on long-term trends in the global marine fishery. *Nat. Commun.* **8**, 14884 (2017).
79. Carozza, D. A., Bianchi, D. & Galbraith, E. D. Formulation, general features and global calibration of a bioenergetically-constrained fishery model. *PLoS One* **12**, 1–28 (2017).
80. Christensen, V. *et al.* The global ocean is an ecosystem: simulating marine life and fisheries. *Glob. Ecol. Biogeogr.* **24**, 507–517 (2015).
81. Christensen, V. & Walters, C. J. Ecopath with Ecosim: methods, capabilities and limitations. *Ecol. Modell.* **172**, 109–139 (2004).
82. Coll, M. *et al.* Advancing global ecological modelling capabilities to simulate future trajectories of change in marine ecosystems. *Front. Mar. Sci.* **Submitted**, (2020).
83. Lam, V. W. Y., Cheung, W. W. L., Reygondeau, G. & Sumaila, U. R. Projected change in global fisheries revenues under climate change. *Sci. Rep.* **6**, 32607 (2016).
84. Cheung, W. W. L., Dunne, J., Sarmiento, J. L. & Pauly, D. Integrating ecophysiology and plankton dynamics into projected maximum fisheries catch potential under climate change in the Northeast Atlantic. *ICES J. Mar. Sci.* **68**, 1008–1018 (2011).
85. UNEP. *Emissions Gap Report 2019. Emissions Gap Report 2019* <https://wedocs.unep.org/bitstream/handle/20.500.11822/30797/EGR2019.pdf?sequence=1&isAllowed=y> (2019).
86. Venter, O. *et al.* Bias in protected-area location and its effects on long-term aspirations of biodiversity conventions. *Conserv. Biol.* **32**, 127–134 (2018).
87. Joppa, L. N. & Pfaff, A. High and far: Biases in the location of protected areas. *PLoS One* **4**, 1–6 (2009).

88. Allan, J. R., Possingham, H. P., Venter, O., Biggs, D. & Watson, J. E. M. B. T.-R. M. in E. S. and E. S. The Extraordinary Value of Wilderness Areas in the Anthropocene. in (Elsevier, 2020). doi:https://doi.org/10.1016/B978-0-12-409548-9.12427-3.
89. Di Marco, M., Ferrier, S., Harwood, T. D., Hoskins, A. J. & Watson, J. E. M. Wilderness areas halve the extinction risk of terrestrial biodiversity. *Nature* **573**, 582–585 (2019).
90. von Lampe, M. *et al.* Why do global long-term scenarios for agriculture differ? An overview of the AgMIP Global Economic Model Intercomparison. *Agric. Econ.* **45**, 3–20 (2014).
91. Higgins, J. Economic impacts of the cod moratorium. *Newfoundland and Labrador Heritage Web Site* (2008).
92. Watson, R. A. *et al.* Global marine yield halved as fishing intensity redoubles. *Fish Fish.* **14**, 493–503 (2013).
93. Bostock, J. *et al.* Aquaculture: Global status and trends. *Philos. Trans. R. Soc. B Biol. Sci.* **365**, 2897–2912 (2010).
94. Costanza, R. *et al.* Changes in the global value of ecosystem services. *Glob. Environ. Chang.* **26**, 152–158 (2014).
95. Costanza, R. *et al.* The value of the world's ecosystem services and natural capital. *Nature* **387**, 253–260 (1997).
96. de Groot, R. *et al.* Global estimates of the value of ecosystems and their services in monetary units. *Ecosyst. Serv.* **1**, 50–61 (2012).
97. Binner, A. *et al.* *Valuing the social and environmental contribution of woodlands and trees in England, Scotland and Wales/ Report to the Forestry Commission (CFSTEN2/14 and CFS 8/17).* (2018).
98. Bateman, I. J. *et al.* Bringing ecosystem services into economic decision-making: land use in the United Kingdom. *Science (80-.).* **341**, 45–50 (2013).
99. Global Preparedness Monitoring Board. *A world at risk. Annual report on global preparedness for health emergencies.* (2019) doi:10.2307/40153772.
100. Bovarnick, A., Baca, J. F., Galindo, J. & Negret, H. Financial Sustainability of Protected Areas in Latin America and the Caribbean : Investment Policy Guidance. *Undp* 162 (2010).
101. Gill, D. A. *et al.* Capacity shortfalls hinder the performance of marine protected areas globally. *Nature* **543**, 665–669 (2017).
102. Coad, L. *et al.* Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. *Front. Ecol. Environ.* **17**, 259–264 (2019).
103. Naidoo, R., Fisher, B., Manica, A. & Balmford, A. Estimating economic losses to tourism in Africa from the illegal killing of elephants. *Nat. Commun.* **7**, 13379 (2016).
104. Karanth, K., DeFries, R., Srivastha, A. & Sankaraman, V. Wildlife tourists in India's emerging economy: potential for a conservation constituency? *Oryx* **46**, 382–390 (2012).
105. Strelecek, F., Lososova, J. & Zdenek, R. The relations between the rent and price of agricultural land in the EU countries. *Agric. Econ. (ZemDdDlská Ekon.* **56**, 558–568 (2018).
106. Mueller, N. D. *et al.* Closing yield gaps through nutrient and water management. *Nature* **490**, 254–257 (2012).
107. Foley, J. A. *et al.* Solutions for a cultivated planet. *Nature* **478**, 337–42 (2011).
108. Chan, K. M. A. *et al.* When agendas collide: Human welfare and biological conservation. *Conserv. Biol.* **21**, 59–68 (2007).
109. Coad, L., Campbell, A., Miles, L. & Humphries, K. *The costs and benefits of forest protected areas for local livelihoods : a review of the current literature.* (2008).
110. Ferraro, P. J. The local costs of establishing protected areas in low-income nations: Ranomafana National Park, Madagascar. *Ecol. Econ.* **43**, 261–275 (2002).
111. Green, J. M. H. *et al.* Local costs of conservation exceed those borne by the global majority. *Glob. Ecol. Conserv.* **14**, e00385 (2018).

112. Guthiga, P. *Opportunity costs of conservation: The case of protected area management in the Kakamega Forest, Western Kenya*. <http://www.jstor.org/stable/resrep14824> (2009).
113. Ward, C., Stringer, L. C. & Holmes, G. Protected area co-management and perceived livelihood impacts. *J. Environ. Manage.* **228**, 1-12 (2018).
114. Ravenelle, J. & Nyhus, P. J. Global patterns and trends in human-wildlife conflict compensation. *Conserv. Biol.* **31**, 1247-1256 (2017).
115. Tauli-Corpuz, V., Alcorn, J., Molnar, A., Healy, C. & Barrow, E. Cornered by PAs: Adopting rights-based approaches to enable cost-effective conservation and climate action. *World Dev.* **130**, 104923 (2020).
116. Tauli-Corpuz, V., Alcorn, J. & Molnar, A. Cornered by Protected Areas. 1-14 (2018).
117. James, A. N., Gaston, K. J. & Balmford, A. Balancing the Earth's accounts. *Nature* **401**, 323-324 (1999).
118. European Commission Environment. Financing Natura 2000. (2020).
119. Lusk, G. Fixing our national parks' funding crisis may mean some sites must close. *MarketWatch* (2019).
120. Lessmann, J., Fajardo, J., Bonaccorso, E. & Bruner, A. Cost-effective protection of biodiversity in the western Amazon. *Biol. Conserv.* **235**, 250-259 (2019).
121. Sangha, K. K., Gerritsen, R. & Russell-Smith, J. Repurposing government expenditure for enhancing Indigenous well-being in Australia: A scenario analysis for a new paradigm. *Econ. Anal. Policy* **63**, 75-91 (2019).
122. Brown, C. *et al.* The cost of enforcing marine protected areas to achieve ecological targets. *Biol. Conserv.* **227**, 259-265 (2018).
123. Stern, N. *The economics of climate change: the Stern review*. (2007).
124. DeFries, R. *et al.* The missing economic risks in assessments of climate change impacts. 1-15 (2019).
125. CBD. Convention on Biological Diversity. www.cbd.int/convention/convention.shtml.
126. UN. *Paris Agreement*. (2015).
127. Li, Y., Mei, B. & Linhares-Juvenal, T. The economic contribution of the world's forest sector. *For. Policy Econ.* **100**, 236-253 (2019).
128. UNWTO. *Tourism highlights 2008 Edition*. (2008).
129. Buckley, R. *et al.* Economic value of protected areas via visitor mental health. *Nat. Commun.* **10**, 5005 (2019).
130. Antonelli, M., Barbieri, G. & Donelli, D. Effects of forest bathing (shinrin-yoku) on levels of cortisol as a stress biomarker: a systematic review and meta-analysis. *Int. J. Biometeorol.* **63**, 1117-1134 (2019).
131. Gao, J. How China will protect one-quarter of its land. *Nature* **569**, 457 (2019).
132. Donofrio, S., Maguire, P., Merry, W. & Zwick, S. *Financing emissions reductions for the future. State of the voluntary carbon markets 2019*. (2019).
133. Waldron, A. *et al.* Reductions in global biodiversity loss predicted from conservation spending. *Nature* **551**, 364-367 (2017).
134. Cohen, N., Robbins, P. & Denault, J.-F. *A guide to social return on investment. Green Business: An A-to-Z Guide* (2012) doi:10.4135/9781412973793.n132.
135. Arvidson, M., Lyon, F., McKay, S. & Moro, D. Valuing the social? The nature and controversies of measuring social return on investment (SROI). *Volunt. Sect. Rev.* **4**, 3-18 (2013).
136. Eagles, P., McCool, S. & Haynes, C. Sustainable tourism in protected areas. Guidelines for planning and management. (2002).
137. Buckley, R. Sustainable tourism: Research and reality. *Ann. Tour. Res.* **39**, 528-546 (2012).

138. Spenceley, A. *et al.* Visitor management. in *Protected area governance and management*. (eds. Worboys, G., Lockwood, M., Kothari, A., Feary, S. & Pulsford, I.) 715–750 (ANU Press, Canberra, 2015).
139. Gurney, G. G., Pressey, R. L., Cinner, J. E., Pollnac, R. & Campbell, S. J. Integrated conservation and development : evaluating a community-based marine protected area project for equality of socioeconomic impacts. *Philos. Trans. R. Soc. London B* **370**, 20140277 (2015).
140. Ingram, D. J. *et al.* Indicators for wild animal offtake: Methods and case study for African mammals and birds. *Ecol. Soc.* **20**, 40 (2015).
141. Chapman, C. A. *et al.* Providing health care to improve community perceptions of protected areas. *Oryx J. Fauna Preserv. Soc.* **49**, 636–642 (2015).
142. Hayes, T. M. Parks, People, and Forest Protection: An Institutional Assessment of the Effectiveness of Protected Areas. *World Dev.* **34**, 2064–2075 (2006).
143. Spenceley, A., Snyman, S. & Rylance, A. Revenue sharing from tourism in terrestrial African protected areas. *J. Sustain. Tour.* **27**, 720–734 (2019).
144. Snyman, S. & Bricker, K. S. Living on the edge: benefit-sharing from protected area tourism. *J. Sustain. Tour.* **27**, 705–719 (2019).
145. Bedelian, C. *Conservation and ecotourism on privatised land in the Mara, Kenya: The case of conservancy land leases. LDPI Working Paper 9* (2012).
146. Wood, S. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. *J. R. Stat. Soc. B* **73**, 3–36 (2011).
147. Burnham, K. P. & Anderson, D. R. *Model selection and multimodel inference: a practical information-theoretic approach*. (Springer-Verlag, 2002).
148. Schägner, J. P. *et al.* Monitoring recreation across European nature areas: A geo-database of visitor counts, a review of literature and a call for a visitor counting reporting standard. *J. Outdoor Recreat. Tour.* **18**, 44–55 (2017).
149. Schägner, J. P., Brander, L., Maes, J., Paracchini, M. L. & Hartje, V. Mapping recreational visits and values of European National Parks by combining statistical modelling and unit value transfer. *J. Nat. Conserv.* **31**, 71–84 (2016).
150. Toivonen, T. *et al.* Social media data for conservation science: A methodological overview. *Biol. Conserv.* **233**, 298–315 (2019).
151. Tenkanen, H. *et al.* Instagram, Flickr, or Twitter: Assessing the usability of social media data for visitor monitoring in protected areas. *Sci. Rep.* **7**, 17615 (2017).
152. OMT. *Tourism Towards 2030 Global Overview. UNTWO General Assembly, 19th Session* (2011).
153. Balmford, A. *et al.* A global perspective on trends in nature-based tourism. *PLoS Biol.* **7**, e1000144–e1000144 (2009).
154. UNWTO. *World Tourism Barometer 2019*. (2019).
155. Balmford, A. *et al.* Walk on the Wild Side: Estimating the Global Magnitude of Visits to Protected Areas. *PLoS Biol.* **13**, (2015).
156. Spalding, M. *et al.* Mapping the global value and distribution of coral reef tourism. *Mar. Policy* **82**, 104–113 (2017).
157. Fredman, P. & Wikström, D. Income elasticity of demand for tourism at Fulufjället National Park. *Tour. Econ.* **24**, 51–63 (2018).
158. Weiler, S. A park by any other name: National Park designation as a natural experiment in signaling. *J. Urban Econ.* **60**, 96–106 (2006).
159. Fredman, P., Friberg, L. H. & Emmelin, L. Increased Visitation from National Park Designation. *Curr. Issues Tour.* **10**, 87–95 (2007).

160. OECD.Stat. Creditor Reporting System.
161. Miller, D. C. Explaining global patterns of international aid for linked biodiversity conservation and development. *World Dev.* **59**, 341-359 (2014).
162. Mamun, A., Martin, W. & Tokgoz, S. *Reforming agricultural support for improved environmental outcomes. IFPRI Discussion Paper 1891* <http://ebrary.ifpri.org/cdm/ref/collection/p15738coll2/id/133527> (2019) doi:10.2499/p15738coll2.133527.
163. OECD. *Reforming agricultural subsidies to support biodiversity in Switzerland*. <https://www.oecd.org/environment/resources/Policy-Paper-Reforming-agricultural-subsidies-to-support-biodiversity-in-Switzerland.pdf> (2017).
164. Stokstad, E. United Kingdom to embark on 'agricultural revolution' in break from EU farm subsidies. *Science* (80-.). **Jan 17**, (2020).
165. Baylis, K., Peplow, S., Rausser, G. & Simon, L. Agri-environmental policies in the EU and United States: A comparison. *Ecol. Econ.* **65**, 753-764 (2008).
166. FAO. FAO statistical databases. www.faostat.fao.org.
167. OECD-DAC. *Aid to Agriculture and Rural Development*. (2015) doi:10.3362/9781780443003.001.
168. Blasiak, R. & Wabnitz, C. C. C. Aligning fisheries aid with international development targets and goals. *Mar. Policy* **88**, 86-92 (2018).
169. Brander, L. *et al.* Scaling-up ecosystem service values : Using GIS and meta-analysis for value transfer. (2011).
170. Carrasco, L. R., Webb, E. L., Symes, W. S., Koh, L. P. & Sodhi, N. S. Global economic trade-offs between wild nature and tropical agriculture. *PLoS Biol.* **15**, 1-22 (2017).
171. Hussain, S. *et al.* *The Economics of Ecosystems and Biodiversity Quantitative Assessment. Final report to UNEP*. (2011).
172. Ghermandi, A., van den Bergh, J. C. J. M., Brander, L. M., de Groot, H. L. F. & Nunes, P. A. L. D. Values of natural and human-made wetlands: A meta-analysis. *Water Resour. Res.* **46**, (2010).
173. Chaikumbung, M., Doucouliagos, H. & Scarborough, H. The economic value of wetlands in developing countries: A meta-regression analysis. *Ecol. Econ.* **124**, 164-174 (2016).
174. Rao, N. S., Ghermandi, A., Portela, R. & Wang, X. Global values of coastal ecosystem services: A spatial economic analysis of shoreline protection values. *Ecosyst. Serv.* **11**, 95-105 (2015).
175. Harris, D. L. *et al.* Coral reef structural complexity provides important coastal protection from waves under rising sea levels. *Sci. Adv.* **4**, eaao4350 (2018).
176. Roche, R. C. *et al.* Recreational Diving Impacts on Coral Reefs and the Adoption of Environmentally Responsible Practices within the SCUBA Diving Industry. *Environ. Manage.* **58**, 107-116 (2016).
177. Nordhaus, W. D. Revisiting the social cost of carbon. *Proc. Natl. Acad. Sci.* **114**, 1518 LP - 1523 (2017).
178. Creedy, J. & Passi, H. *Public Sector Discount Rates: A Comparison of Alternative Approaches. Australian Economic Review* vol. 51 (2018).
179. Park, S. Optimal Discount Rates for Government Projects. *ISRN Econ.* **2012**, 1-13 (2012).
180. Balmford, A., Gaston, K. J., Blyth, S., James, A. & Kapos, V. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *PNAS* **100**, 1046-1050 (2003).
181. Bruner, A. G., Gullison, R. E. & Balmford, A. Financial Costs and Shortfalls of Managing and Expanding Protected-Area Systems in Developing Countries. *Bioscience* **54**, 1119-1126 (2004).
182. Balmford, A., Gravestock, P., Hockley, N., McClean, C. J. & Roberts, C. M. The worldwide costs of marine protected areas. *Proc. Natl. Acad. Sci. U. S. A.* **101**, 9694-7 (2004).

183. Geldmann, J., Joppa, L. N. & Burgess, N. D. Mapping Change in Human Pressure Globally on Land and within Protected Areas. *Conserv. Biol.* **28**, 1604–1616 (2014).
184. Venter, O. *et al.* Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nat. Commun.* **7**, 1–11 (2016).
185. James, A. N., Green, M. J. B. & Paine, J. R. *A Global Review of Protected Area Budgets and Staffing.* (World Conservation Press, Cambridge, UK, 1999).
186. Green, J. M. H. *et al.* Estimating management costs of protected areas: A novel approach from the Eastern Arc Mountains, Tanzania. *Biol. Conserv.* **150**, 5–14 (2012).
187. Waldron, A. *et al.* Targeting global conservation funding to limit immediate biodiversity declines. *PNAS* **110**, 12144–12148 (2013).
188. International Bank for Reconstruction and Development and World Bank. *Global Purchasing Power Parities and Real Expenditures. 2005 International Comparison Programme.* (2008).
189. Sanderson, E. W. *et al.* The Human Footprint and the Last of the Wild: The human footprint is a global map of human influence on the land surface, which suggests that human beings are stewards of nature, whether we like it or not. *Bioscience* **52**, 891–904 (2002).
190. Naidoo, R. *et al.* Integrating economic costs into conservation planning. **21**, 681–687 (2006).
191. Kaufmann, D., Kraay, A. & Mastruzzi, M. *The Worldwide Governance Indicators Methodology and Analytical Issues.* (2010).
192. Miller, D. C., Agrawal, A. & Timmons Roberts, J. Biodiversity, governance, and the allocation of international aid for conservation. *Conserv. Lett.* **6**, 12–20 (2013).
193. Binet, T., Diazabakana, A. & Hernandez, S. *Sustainable financing of Marine Protected Areas in the Mediterranean: a financial analysis.* (2015) doi:10.1787/9789264276208-7-en.
194. Watson, R. A. A database of global marine commercial, small-scale, illegal and unreported fisheries catch 1950–2014. *Sci. Data* **4**, 170039 (2017).
195. Watson, R. A. & Tidd, A. Mapping nearly a century and a half of global marine fishing: 1869–2015. *Mar. Policy* **93**, 171–177 (2018).
196. Gaffin, S. R., Xing, X. & Yetman, G. Country-Level GDP and Downscaled Projections Based on the SRES A1, A2, B1, and B2 Marker Scenarios, 1990–2100. (2002).
197. Gaffin, S. R., Rosenzweig, C., Xing, X. & Yetman, G. Downscaling and Geo-spatial Gridding of Socio-economic Projections from the IPCC Special Report on Emissions Scenarios (SRES). *Glob. Environ. Chang.* **14**, 105–123 (2004).
198. Adams, V. M., Segan, D. B. & Pressey, R. L. How much does it cost to expand a protected area system? some critical determining factors and ranges of costs for Queensland. *PLoS One* **6**, (2011).